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Use of wetland buffer areas to reduce nitrogen transport from forested catchments: Retention capacity, emissions of N_2O and CH_4 and vegetation composition dynamics

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Academic dissertation

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ABSTRACT

The use of buffer areas in forested catchments has been actively researched during the last 15 years; but until now, the research has mainly concentrated on the reduction of sediment and phosphorus loads, instead of nitrogen (N). The aim of this thesis was to examine the use of wetland buffer areas to reduce the nitrogen transport in forested catchments and to investigate the environmental impacts involved in their use. Besides the retention capacity, particular attention was paid to the main factors contributing to the N retention, the potential for increased N_2O emissions after large N loading, the effects of peatland restoration for use as buffer areas on CH_4 emissions, as well as the vegetation composition dynamics induced by the use of peatlands as buffer areas.

To study the capacity of buffer areas to reduce N transport in forested catchments, we first used large artificial loadings of N, and then studied the capacity of buffer areas to reduce ammonium (NH_4-N) export originating from ditch network maintenance areas in forested catchments. The potential for increased N_2O emissions were studied using the closed chamber technique and a large artificial N loading at five buffer areas. Sampling for CH_4 emissions and methane-cycling microbial populations were done on three restored buffer areas and on three buffers constructed on natural peatlands. Vegetation composition dynamics was studied at three buffer areas between 1996 and 2009.

Wetland buffer areas were efficient in retaining inorganic N from inflow. The key factors contributing to the retention were the size and the length of the buffer, the hydrological loading and the rate of nutrient loading. Our results show that although the N_2O emissions may increase temporarily to very high levels after a large N loading into the buffer area, the buffer areas in forested catchments should be viewed as insignificant sources of N_2O. CH_4 fluxes were substantially higher from buffers constructed on natural peatlands than from the restored buffer areas, probably because of the slow recovery of methanogens after restoration. The use of peatlands as buffer areas was followed by clear changes in plant species composition and the largest changes occurred in the upstream parts of the buffer areas and the wet lawn-level surfaces, where the contact between the vegetation and the through-flow waters was closer than for the downstream parts and dry hummock sites. The changes in the plant species composition may be an undesired phenomenon especially in the case of the mires representing endangered mire site types, and therefore the construction of new buffer areas should be primarily directed into drained peatland areas.

Keywords: ammonium nitrogen, denitrification, mire vegetation, nitrate nitrogen, peat soil, water protection

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Helsinki, September 2011
Anu Hynninen

LIST OF ORIGINAL PUBLICATIONS

This thesis is based on the following original publications that are referred to in the text by their Roman numerals. Articles **I-III** and **V** are reproduced with the kind permission from the publishers, while the study **IV** is the author version of the submitted manuscript.

I Vikman, A., Sarkkola, S., Koivusalo, H., Sallantaus, T., Laine, J., Silvan, N., Nousiainen, H. & Nieminen, M. 2010. Nitrogen retention by peatland buffer areas at six forested catchments in southern and central Finland. Hydrobiologia 641: 171–183.
doi: 10.1007/s10750-009-0079-0.

II Hynninen, A., Sarkkola, S., Lauren, A., Koivusalo, H. & Nieminen, M. 2011. Capacity of natural and restored peatland buffer areas in reducing ammonium export originating from ditch network maintenance areas in peatlands drained for forestry. Boreal Environment Research 16: 430–440.
http://www.borenv.net/

III Hynninen, A., Fritze, H., Sarkkola, S., Kitunen, V., Nousiainen, H., Silvan, N., Laine, J., Tervahauta, A. & Nieminen, M. 2011. N_2O fluxes from peatland buffer areas under high N loads in five forested catchments in Finland. Wetlands. In press.
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IV Juottonen, H., Hynninen, A., Nieminen, M., Tuomivirta, T, Tuittila, E-S., Nousiainen, H., Yrjälä, K., Tervahauta, A. & Fritze, H. 2011. Methane-cycling microbial communities and methane emission in restored and natural peatland buffer areas. Manuscript.

V Hynninen, A., Hamberg, L., Nousiainen, H., Korpela, L., Nieminen, M. 2011. Vegetation composition dynamics in peatlands used as buffer areas in forested catchments in southern and central Finland. Plant Ecology 212: 1803-1818.
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THE AUTHOR´S CONTRIBUTION TO THE PUBLICATIONS

The author participated in the planning of all the articles together with the co-authors. She organized the field work of the articles **I**, **III**, **IV** and **V**, conducted the field work of the articles **I**, **III** and **IV**, and did the laboratory work of the articles **I** and **III**, and partially in **IV**. The author was responsible for the data analysis of the papers **I**, **II**, **III** and **V** and for model construction in paper **II**. She participated in the writing of the paper **IV** and was fully responsible for the papers **I**, **II**, **III** and **V** as the corresponding author.

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1 INTRODUCTION

1.1 The use of peatland buffer areas in forestry

Forestry operations, such as fertilization (Lundin and Bergqvist 1985), drainage of peat soils (Lundin and Bergquist 1990, Manninen et al. 1998), and harvesting (Nieminen 2003, 2004, Uusivuori et al. 2008) may significantly increase the transport of sediments and nutrients to recipient watercourses. One means of managing water quality in forested catchments is to construct buffer areas between forestry land and recipient water bodies. Buffer areas can be constructed by either simply conducting discharge waters from forested areas to pristine mires or to paludified mineral soils. However, because over half of the peatlands in Finland have been drained, buffer wetlands are very often created by restoring and rewetting sections of drained peatlands by filling in or blocking the main drainage ditches. The use of peatland buffer areas is presently recommended as an effective method to reduce the transport of sediment and nutrients from forested areas (Silvan et al. 2004a, Nieminen et al. 2005, Väänänen et al. 2008). The even topography, the dense moss cover and many favorable physical, chemical and biological properties of peat enable a considerable buffering potential and efficient nutrient and sediment retention (Vasander et al. 2003). However, there are some possible environmental impacts involved in the use of buffer areas. One concern, raised in connection with the use of peatland buffer areas, is that they may enhance the emissions of nitrous oxide (N_2O) and methane (CH_4), which are potent greenhouse gases (Khalil 1999) involved in the global warming and atmospheric deterioration processes (Crutzen 1970). The use of natural mires as buffer areas may induce changes in the plant species composition, which may be an undesirable phenomenon especially in the case of the mires representing endangered mire site types. Also, the peatland buffers created by restoring drained peatlands may release nutrients to through-flow waters during the first few years after restoration operations, such as ditch blocking and tree stand harvesting (Kuuluvainen et al. 2002, Vasander et al. 2003).

1.2 Nitrogen retention capacity of buffer areas in forested catchments

1.2.1 Nitrogen loading from forested catchments

Nitrogen (N) and phosphorus (P) are limiting nutrient resources for plant and microbial growth in most boreal waters (Vitousek et al. 1997, Pietiläinen and Räike 1999, Bergström et al. 2005). An excess N and P input into watercourses may lead to nutrient enrichment, also known as eutrophication, which is a common environmental problem in Finnish inland waters and the coastal areas of the Baltic Sea (Pietiläinen and Räike 1999, Ministry of the Environment 2007). Eutrophication has substantial effects on ecosystem function and composition, including algae blooming and water quality deterioration resulting in changes in the aquatic flora and fauna (Vitousek et al. 1997, Pietiläinen and Räike 1999, Ministry of the Environment 2007). Abundance of phytoplankton may increase creating surface accumulations and decreasing visibility, thereupon reducing the colonization depth of macroalgae and seagrasses (HELCOM 2009). An increased amount of sedimented and degrading algae consume oxygen (Vitousek et al. 1997), and the development of the anoxic conditions may then lead to an excess release of P from the bottom sediments (Kauppila

and Bäck 2001). Thus, a reduction in P and N export to inland waters, coastal waters, and groundwater has been one of the key interests in the Finnish decision-in-principle "Water Protection Policy Outlines to 2015" (Ministry of the Environment 2007).

Presently, about 6% of the total N loading from Finland into the Baltic Sea originates from forestry operations, while the contribution of agriculture is about 80% (Nyroos et al. 2006, HELCOM 2009). Although the contribution of forestry operations is not large at the national scale, the effects of forestry can locally be very important. Forestry is often practiced in distant areas where other anthropogenic N sources are low and besides, the nutrient loadings from forestry operations may occasionally be high. While the average background leaching of ammonium (NH_4-N) and nitrate (NO_3-N) from unmanaged forested catchments in southern Finland is about 300 g ha^{-1} a^{-1} (Kortelainen and Saukkonen 1998), fertilizations with N in mineral soil sites may cause an excess leaching of several kilograms per hectare during the first few years after application (Lundin and Bergquist 1985). In peatlands, the effects of forest operations on nutrient export are often even larger than for mineral soil sites. For example, clear-cutting of Norway spruce dominated stands on fertile peatland sites may increase the N export by about 4 kg ha^{-1} a^{-1} (Uusivuori et al. 2008) and P concentrations in ditch outflow may increase from a level of less than 20 to over 500 µg l^{-1} during the first 2–3 years after harvesting of drained, infertile Scots pine stands (Nieminen 2003). Ditch network maintenance operations on forest land may result in manifold increases in the loadings of inorganic N, especially ammonium N (Manninen et al. 1998, Joensuu et al. 2002).The release of nutrients is usually largest during 1-3 years after the operation; however, high nutrient loadings can occur even 10 years later. In the near future, ditch network maintenance, fertilizations and energy wood harvesting are expected to increase and also the N loadings from forested catchments are expected to grow (Ministry of the Environment 2007). Therefore the control on the effects of forestry operations on watercourses will become increasingly important.

1.2.2 The use of wetland buffer areas in reducing nutrient transport

It is currently recommended to use buffer areas in reducing the nutrient export from forested areas to watercourses (Nieminen et al. 2005, Väänänen et al. 2006, 2008). The use of buffer areas in filtering nitrogen from discharging waters has been actively researched from the viewpoint of wastewater management in municipalities and industries (Surakka and Kämppi 1971, Boyt et al. 1977, Sloey et al. 1978, Nichols 1983, Kent 1987, Tanner et al. 1994), as well as in peat production areas (Ihme et al. 1991, Huttunen et al. 1996) and in agriculture (Peterjohn and Correll 1984, Mander et al. 1997, Woltemade 2000, Dosskey 2001). Surakka and Kämppi (1971) reported an average N removal efficiency of 62% for a municipal wastewater loaded buffer area, which was created on a drained peatland in Eastern Finland. Kent (1987) reported a somewhat higher N retention efficiency (>80%) for a wastewater loaded marsh wetland in Canada. In peat production areas, use of buffers have also proven to be effective; a buffer area covering 4.8% of the catchment and created on a pristine mire in northern Finland reduced the N concentrations by about 40% (Huttunen et al. 1996) and three buffers on pristine mires in northern Finland covering 1.5–4.8% of catchment area reduced N transport by 38–74% (Ihme et al. 1991). In addition, buffers have significantly reduced the N loading from agricultural fields (Gilliam et al. 1997, Mander et al. 1997, Woltemade 2000, Dosskey 2001). Two riparian buffer areas with grey alder stands in Estonia reduced the N loading originating from agricultural fields by about 80% (Mander et al. 1997) and a riparian forested buffer in Maryland by about 89% (Peterjohn and Correll

1984). Consequently, buffer areas appear to be capable of effectively reducing N loadings from different pollution sources and under different site and environmental conditions.

In forestry, the research has mainly focused on whether buffer areas can be used to reduce the loadings of suspended solids (SS) and P (Nieminen et al. 2005, Väänänen et al. 2006, 2008). In these studies, large buffer areas (relative size >1%) were efficient in reducing high loadings, with retention capacities of >90% for P and of 70–100% for SS (Nieminen et al. 2005, Väänänen et al. 2008). The efficiency of smaller buffer areas (relative size <1%) was lower, with the reduction efficiencies of 20–90% for P loadings and 50–60% for SS (Nieminen et al. 2005, Väänänen et al. 2008). However, less attention has been paid to the N retention capacity of the buffer areas constructed in forestry. The few previous studies that have examined the N retention in boreal forested areas have either been conducted using an exceptionally large buffer area from the viewpoint of operational forestry (Silvan et al. 2003, 2004a) or then the buffer areas have been subjected to an N loading that is not higher than the background loading from undisturbed forest areas (Sallantaus et al. 1998, Lundin et al. 2008). Information on the N retention capacity of buffer areas is also important, since in boreal areas with low atmospheric N deposition levels, N may be even more limiting nutrient for phytoplankton in lakes than P (Bergström et al. 2005).

1.2.3 The key-factors controlling the retention capacity of peatland buffer areas

The N retention capacity of the buffer areas is controlled by the physical, chemical and biological characteristics of the buffer area and the upstream catchment area (**Fig. 1**). The hydrological loading entering the buffer area is considered to be one of the key factors controlling the retention capacity (Correll 1997, Gilliam et al. 1997, Woltemade 2000). During high flow episodes the water residence time is short and the formation of continuous flow channels across the buffer area decreases the retention efficiency (Woltemade 2000, Väänänen et al. 2006, 2008, Ronkanen and Kløve 2009) **(Fig. 2)**. Even if the buffers in such conditions may retain sediment and the nutrients adhered to solids, dissolved nutrients are not retained effectively (Woltemade 2000, Dosskey 2001). Under low flow conditions, the contact time between through-flowing water and the nutrient sinks in soil and vegetation is longer and the retention of dissolved nutrients is more effective (Heikkinen et al. 1994, Sallantaus et al. 1998, Dosskey 2001, Väänänen et al. 2008).

The retention capacity of the buffer areas is also strongly related to their relative size, i.e. the size of the buffer relative to the size of the upstream catchment area. Buffer areas covering an area larger than 1% of the catchment area have been proven to be effective in water purification, while in smaller buffer areas, the short water residence time may significantly decrease their retention capacity (Sallantaus et al. 1998, Woltemade 2000, Liljaniemi et al. 2003, Nieminen et al. 2005). The large size itself is a contributing factor for nutrient retention, because the nutrient sinks are correspondingly larger, which results in lower relative loading and lower probability of saturation of the nutrient sinks.

The pattern and duration of N loading may considerably affect the retention capacity of buffer areas (Correll 1997, Ronkanen and Kløve 2009). Although buffer areas may effectively reduce the transport of nutrients under increased loadings (Correll 1997, Silvan et al. 2003, 2004a, Väänänen et al. 2008), the retention efficiency may decrease when the loading is at a very high level (Ronkanen and Kløve 2009). The decreased retention efficiency under high N loadings is often associated with the concurrent large hydrological

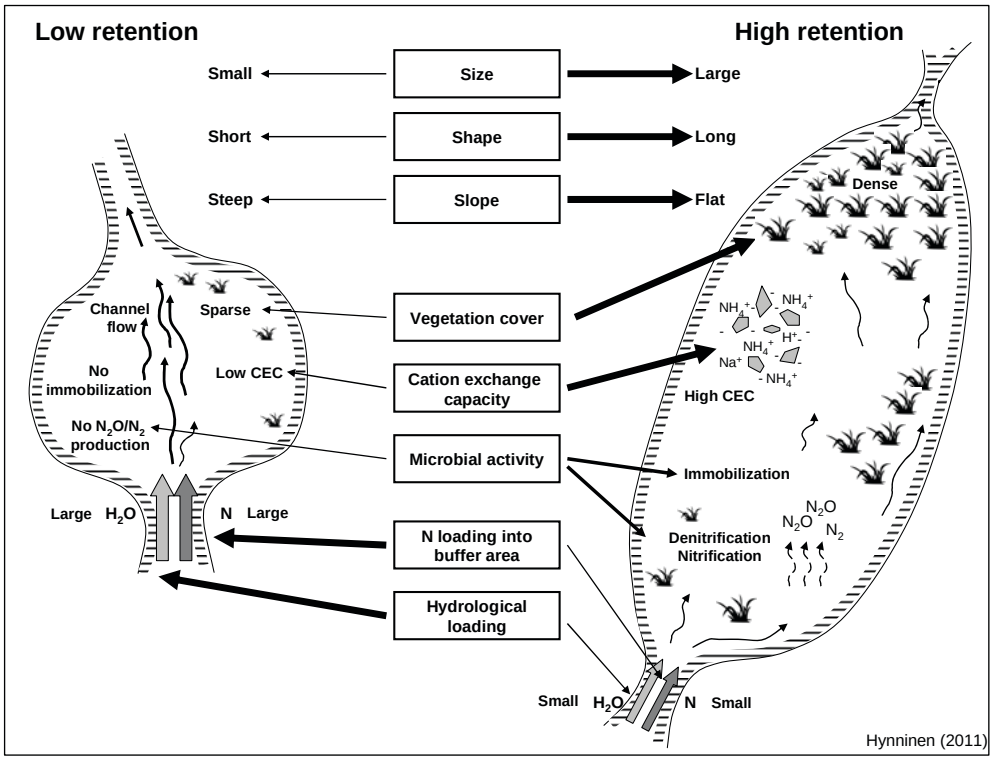

Figure 1. The main factors contributing to the retention capacity of buffer areas.

loadings, which may lead to the canalization of water flow (Ronkanen and Kløve 2009). Also, when the N input into the buffer area is high, the N sinks in the soil and vegetation may become saturated. The saturation of the N sinks is, however, unlikely to be an equally important factor in forested catchments as in agricultural areas (Bernot et al. 2006, Dorioz et al. 2006) and in the buffers used for waste water treatment (Sloey et al. 1978, Nichols 1983, Ronkanen and Kløve 2009). When the N loading is close to the background levels of forested areas, buffer areas have little effect on through-flow N concentrations or they may even act as N sources to recipient water courses (Liljaniemi et al. 2003, Nieminen et al. 2005, Lundin et al. 2008). A negative retention capacity is a common phenomenon on recently restored peatland buffers, which may release nutrients to through-flow waters during the first few years after restoration operations, such as ditch blocking and tree stand harvesting (Vasander et al. 2003).

Vegetation acts as a sink for N, thus vegetation type and density may affect the nitrogen retention efficiency (Heikkinen et al. 1994, Correll 1997, Kallner Bastviken et al. 2009). A dense vegetation cover increases the N retention capacity directly by the uptake and conversion of inorganic N into less mobile organic forms (Nichols 1983, Huttunen et al. 1996, Kallner Bastviken et al. 2009), and indirectly by slowing down the water movement through the buffer area. The above-ground parts of vegetation may assimilate N effectively during summer growing season, but some N may be released during the wilting and decay of the vegetation in the autumn (Kallner Bastviken et al. 2009). High nutrient inputs outside the growing season in the boreal region are retained only by soil processes.

The capacity of the soil to retain nutrients varies depending on the physical and chemical soil characteristics, such as the cation exchange capacity (Lance 1972), the soil sorption properties (D´Angelo and Reddy 1994), and the form of nutrient in the through-flowing water (Lance 1972). Ammonium (NH_4^+) can be retained into cation exchange sites of the soil (Lance 1972, Heikkinen et al. 1994), while nitrate (NO_3^-) generally remains in soluble form, unless assimilated by vegetation or microbial communities. Peat soils usually have high cation exchange capacity (CEC), which enables a considerable potential to the retention of NH_4^+ (Heikkinen et al. 1994). The effective cation exchange capacity is generally highest in the peat surface layer (Ronkanen and Kløve 2009), however, water table level fluctuations may affect the CEC (Lance 1972, D´Angelo and Reddy 1994). Flooding of aerobic peat soil may result in NH_4^+ release, because anaerobic bacteria has lower requirements for N, leaving more NH_4^+ available for transport from the soil to the water column (D´Angelo and Reddy 1994). Also, when previously anaerobic soil layers become aerobic, the adsorbed NH_4^+ can be oxidized to NO_3^-, which is then easily leached during next inundation (Lance 1972). Recent studies show that NH_4^+ oxidation may also occur under anaerobic conditions (Mulder et al. 1995).

The growth and activity of the soil micro-organisms is controlled by the availability of energy and nutrients in the peatland ecosystems, thus microbial communities are likely to thrive under a high N inflow into the buffer areas (Heikkinen et al. 1994, Peacock et al.

Figure 2. The formation of flow channels during the high flow episodes may significantly decrease the retention efficiency of a buffer area.

2001, Silvan et al. 2003). A significant amount of N can be immobilized through an increase in the size and the N concentrations of the microbial biomass (Heikkinen et al. 1994, Silvan et al. 2003), but part of the nitrogen assimilated in microbial cells can be released along with dying and decay of microbial biomass (Lance 1972). Microbial communities are responsible for the production of gaseous N_2O and N_2 through nitrification and denitrification, which can account for a substantial proportion of the total N loss from the buffer areas (Gilliam et al. 1997, Silvan et al. 2002). N_2O is highly soluble in water, and therefore some N_2O may also be transported by the runoff water from a peatland area. However, this is only a fraction of what is emitted into the atmosphere (Nieminen 1998).

1.3 Environmental impacts of the use of buffer areas

1.3.1 Possible environmental impacts

Although buffer areas are an efficient method in removing N from discharge waters from forested catchments, some negative impacts may be involved. One such impact is that the construction of buffer areas by rewetting and restoring drained peatland sites may initially increase the export of nutrients (Kuuluvainen et al. 2002, Vasander et al. 2003). However, although an enhanced export would occur during the buffer construction and a few years after, buffer areas are likely to turn into nutrient-accumulating systems over time (Liljaniemi et al. 2003). Another concern raised in connection with the use of peatland buffer areas is that they may enhance the emissions of nitrous oxide (N_2O) and methane (CH_4). Also, the plant species composition of rare and endangered mire site types may change because of their use as buffer areas, but limited information is available on the dynamics of plant species composition and the existence of endangered plant species in mires used as buffer areas.

1.3.2 Emissions of the gases methane (CH_4) and nitrous oxide (N_2O)

N_2O and CH_4 as greenhouse gases

Considered over a 100 year period, CH_4 is 23 times and N_2O 296 times more effective in trapping heat in the atmosphere than carbon dioxide (CO_2) (Houghton et al. 2001). Besides the contribution to the global warming, N_2O is involved in the depletion of the stratospheric ozone, the appearance of photochemical smog and the formation of acid rain (Crutzen 1970, Vitousek et al. 1997, Olivier et al. 1998).

N_2O and CH_4 are emitted from a variety of natural and human-influenced sources (Vitousek et al. 1997, Olivier et al. 1998). Presently over half of the CH_4 emissions originate from anthropogenic activities, e.g. agriculture, landfills and the production and the use of fossil fuels (Houghton et al. 2001). For N_2O, the anthropogenic emissions, mainly from agriculture, industrial combustion and transportation, contribute to about 40% of the total (Karttunen et al. 2008). The most important natural sources of CH_4 are the methane-producing bacteria in swamps and wetlands, including peat-forming mires, whereas the largest part of natural N_2O originates from biological sources in soil and water, particularly from microbial action in wet tropical forests (Olivier et al. 1998).

As a result of human activities atmospheric N_2O and CH_4 concentrations have risen since the pre-industrial times about 20% and 150%, respectively (Houghton et al. 2001).

According to estimates N_2O emissions would further grow at 0.2% rate annually in the 21st century and the emissions of CH_4 would grow by about 0.6% (Vuuren et al. 2005).

N_2O emissions from peatland buffer areas

N_2O is formed in soils mainly by two pathways: from nitrification of ammonium (NH_4^+) to nitrate (NO_3^-) in aerobic environments, and from denitrification of nitrate (NO_3^-) to molecular nitrogen (N_2) in anaerobic environments (Patrick and Tushneem 1972, Nichols 1983, Koops et al. 1997). The production of N_2O is related to the amount and activity of soil microbes, which in turn is regulated by the chemical and physical conditions of the soil, such as temperature (Kaiser et al. 1998, Smith et al. 1998, Teiter and Mander 2005, Koponen et al. 2006), pH (Knowles 1982, Klemedtsson et al. 1995), C:N ratio of the soil (Klemedtsson et al. 1995), water table level and oxygen content (Knowles 1982, Kaiser et al. 1998, Smith et al. 1998). A large availability of NH_4^+ and NO_3^- (e.g. Regina 1996, Kaiser et al. 1998, Baggs et al. 2003) and readily-decomposable organic material may enhance the production (Knowles 1982, Kaiser et al. 1998, Baggs et al. 2003). The activity of the micro-organisms is favoured by high soil temperatures and the N_2O emissions from boreal peatlands are therefore highest during the growing season (Knowles 1982, Kaiser et al. 1998, Bedard-Haughn et al. 2003, Pihlatie et al. 2004, Teiter and Mander 2005). Large emissions of N_2O may also occur outside growing season, especially during episodes of freezing and thawing (Kaiser et al. 1998).

Several studies have quantified the emissions of N_2O from natural peatlands and peatlands drained for forestry purposes (Klemedtsson et al. 1995, Martikainen et al. 1995, Regina et al. 1996, Koops et al. 1997, Huttunen et al. 2003a, Von Arnold et al. 2005, Alm et al. 2007). After drainage for forestry, the N_2O emissions are usually high from nitrogen-rich, minerotrophic peatlands, while the drainage of infertile, ombrotrophic peatlands does not necessarily lead to increased N_2O production (Martikainen et al. 1995, Regina et al. 1996). Nitrous oxide emissions are generally low from rewetted and restored wetland ecosystem sites (Höper et al. 2008). Natural peatlands with small N concentrations in the surface peat may even act as weak sinks for N_2O, because a high water table level limits oxygen diffusion into the soil, resulting in low nitrogen mineralization and nitrification rates (Martikainen et al. 1995, Teiter and Mander 2005, Von Arnold et al. 2005, Höper et al. 2008). The hydrological conditions in peatland buffer areas are different from natural and drained peat soils in that the water level is generally clearly above the soil level and the surface waters are in constant movement across the buffer area. As the N inputs to peatland buffer areas can also be larger than those into other types of peat soils, the N_2O emission measured in natural or drained peatlands can not be applied to peatland buffer areas.

Highly increased N loadings into buffer areas may increase the N_2O emissions (Silvan et al. 2002, Hefting et al. 2003). Given that a large proportion of a forested catchment area is harvested or fertilized concurrently; the N input may be considerable, leading to increased N_2O emissions from buffer areas.

CH_4 emissions from peatland buffer areas

Methane (CH_4) emission is a result of the activity of two microbial groups, methanogens and methanotrophs. Methanogenic archaea produce CH_4 in anaerobic conditions, mainly below the soil water table level, whereas methanotrophic α- and γ-proteobacteria are

responsible for the oxidation of CH_4, which occurs in the presence of oxygen (Sundh et al. 1994, Whalen 2005). The distribution of methanotrophs follow the regimes of the methanogens in peatlands, and CH_4 oxidation is the most active in the aerobic layer close to the level of the water table, where the supplies of CH_4 are high (Sundh et al. 1994, Whalen 2005, Basiliko et al. 2007, Larmola et al. 2010).

The temporal and spatial variability in the CH_4 emissions is dependent on the fluctuations of the peat temperatures. A high peat temperature enhances the emissions (Mikkelä et al. 1995, Eriksson et al. 2010), and therefore largest emissions usually occur during summer growing season. High productivity of plants and deep rooting plant species support CH_4 production by providing litter, oxygen and root exudates into the peat layer, and also by offering an effective route for the CH_4 transport through the root-shoot pathway (Tuittila et al. 2000, Knorr et al. 2008, Eriksson et al. 2010).

Natural peatlands can be significant sources of CH_4 into the atmosphere (Huttunen et al. 2003b) due to prevailing anoxic conditions (Limpens et al. 2008) and slow degradation process that provide large amount of substrate for CH_4 production (Glatzel et al. 2004). Drainage of peatlands reduces the emissions, because emerging aerobic conditions suppress the activity of methanogens (Kettunen et al. 1999, Freeman et al. 2002), while concurrently, methanotrophs are not affected much (Roulet et al. 1993, Sundh et al. 1994). The aerobic conditions also lead to enhanced decomposition rates, which is associated with a decrease in the amount of substrate available for CH_4 production (Komulainen et al. 1998, Huttunen et al. 2003b, Basiliko et al. 2007, Eriksson et al. 2010).

Restoration of drained peatlands involves raising the soil water table level, which is gradually followed by an increasing cover of mire plant species and a decreasing cover of forest species. After a successful rewetting carbon cycle typical for mire ecosystem is slowly revitalized. However, the few studies that have assessed CH_4 emissions on restored peatlands indicate that although restoration increases the emissions (Tuittila et al. 2000, Waddington and Day 2007) they remain lower than for pristine mires, at least during the first two-three years after restoration (Komulainen et al. 1998, Tuittila et al. 2000). The reason for the low rate of CH_4 release after rewetting is not fully understood, however, one reason could be the very slow re-establishment of methanogenic bacteria after prolonged aeration (Tuittila et al. 2000). The aeration may have restricted methanogenesis to distant anoxic microenvironments, which can result in large spatial heterogeneity in the methanogenic communities and in the CH_4 emissions after rewetting (Knorr et al. 2008). However, in restored peatlands used previously for peat extraction, CH_4 production and oxidation potentials have recovered in 4–30 years and even exceeded those of natural sites (Glatzel et al. 2004, Basiliko et al. 2007).

Peatland buffer areas can be constructed on natural mires or forestry-drained peatlands that have been restored and rewetted. The effect of restoration of peatlands for use as buffer areas on the CH_4 emissions and the CH_4 cycling microbial communities has not yet been studied.

1.3.3 Changes in vegetation composition in peatland buffer areas

One of the consequences of the use of peatland buffer areas is that the large nutrient and sediment loadings induce changes in the plant species composition and dynamics (Aerts et al. 1995, Vitousek et al. 1997, Bowman and Bilbrough 2001, Saari et al. 2010a,b). The changes may be an undesirable phenomenon especially if the mires used as buffer areas represent endangered site types. It is presently recommended that endangered mire site

types in their natural or nearly natural state should be preserved as habitats of special importance and their management and utilization actions should be carried out in a manner which preserves the special features of the habitats. Provided that the vegetational changes are significant, the use of endangered mire site types as buffer areas should be carefully considered.

Hydrological conditions in peatland buffer areas differ from natural peatlands in that the water level is generally above the soil surface and the surface waters are in constant movement across the buffer area. Nutrients and sediment are effectively transported by the overland flow and therefore the inputs of nutrients and sediment to peatland buffer areas can be significantly larger than those into other types of peat soils (Sloey et al. 1978, Silvan et al. 2004a). The nutrient input to the buffer areas enhances the growth of some plant species and large changes may follow in plant species composition (Aerts et al. 1995, Vitousek et al. 1997, Bowman and Bilbrough 2001, Silvan et al. 2004a). For instance, sedge, graminoid and herb species have been reported to benefit from increased nutrient availability, whereas the cover of dwarf shrubs and *Sphagnum* decrease (Eriksson et al. 2010). On the species level, *Menyanthes trifoliata* and *Carex lasiocarpa* were particularly favoured by the use of the peatland as a buffer area (Huttunen et al. 1996). In a study by Silvan et al. (2004a), increases in the biomasses of sedges, *Sphagnum* and herbs were observed, and especially *Eriophorum vaginatum* benefited from the increased nutrient supply in a peatland buffer area in central Finland.

Sphagnum species respond quickly to increased N loadings with increased uptake of N and increased production rates (Vitt et al. 2003). However, the accumulation of N may soon reach a critical value, and further N additions may even result in a reduction of the *Sphagnum* growth (Gunnarson and Rydin 2000, Berendse et al. 2001, Gunnarson et al. 2004), and they then lose their competitive advantage to vascular plants (Berendse et al. 2001). Finally, as a result of the continuing high nutrient inputs, the plant society may transform into a vascular-plant-dominated habitat (Huttunen at al. 1996, Gunnarson et al. 2004).

In restored and rewetted peatland sites, the success of rewetting and vegetation drainage succession phase at the time of restoration largely control the rate at which vegetation changes take place (Jauhiainen et al. 2002, Höper et al. 2008). The time since the drainage occurred is one of the key factors determining the success of restoration, as restoration more likely promotes the area to attain its original habitat type in recently drained areas than in areas with a long drainage history (Laine et al. 1995, Vasander et al. 2003). If the restoration proceeds successfully, raising water table level is followed by increasing cover of mire species and decreasing cover in forest species. When the mire vegetation becomes better established, the peat and carbon accumulation process starts again (Komulainen et al. 1998, Woltemade 2000). Restoration of mire vegetation may also be dependent on the initial nutrient status of the site, being faster at the more nutrient-rich sites than at the poor sites (Komulainen et al. 1998).

1.4 Aims of the thesis

The overall aim of this thesis was to study the capacity of wetland buffer areas to reduce N loadings from boreal forested catchments in Finland, and to investigate the environmental impacts involved in their use. The more specific aims were:

- To quantify the efficiency of buffer areas in retaining inorganic nitrogen (NO_3-N and NH_4-N) from discharge waters from forested catchments, and to clarify the main factors that contribute to the N retention capacity. Papers I and II.
- To quantify the fluxes of N_2O from peatland buffer areas, and to study whether an increase in N input to buffer areas leads to a concurrent increase in the N_2O emissions. Paper III.
- To investigate if the restoration of peatlands for the use as buffer areas increases the CH_4 emissions and affects the CH_4-cycling microbial populations. Paper IV.
- To investigate the long-term changes in the vegetation composition after the establishment of buffer areas on natural and restored peatlands. Paper V.

2 MATERIALS AND METHODS

2.1 Experimental design

The studies were carried out on eight buffer areas located in forested catchments in south-central Finland **(Table 1, Figs. 3 and 4,** Silvan et al. 2004a**)**. The buffer areas covered an area of 0.1–1.0 hectares, accounting for 0.1–4.9% of the upstream catchment area. Except for one area constructed on paludified mineral soil, the buffer areas were classified as either undrained, natural mires during the buffer construction or they were drained peatlands that were rewetted and restored during 1995–1997. The buffer areas represented different site types and they were located in varying climatic conditions. The soil chemical and physical characteristics of the experimental peatlands are presented in **Table 2.**

Field measurements for water table levels and peat temperatures, and sampling for soil water quality, chemical and physical peat characteristics and emissions of N_2O and CH_4 were made from sampling positions placed systematically on four to seven lines depending on the length of the buffer. The sampling lines were laid over the buffer areas in a perpendicular position to the main direction of the runoff. Every buffer area was equipped with boardwalks in order to minimize the disturbance caused by walking.

The vegetation inventory was made from 27, 52 and 57 plots (2m x 2m) at the three buffer areas (Paper V). The plots were placed systematically on 4–6 transects, which were located at 10 meters intervals at two buffer areas, and at 10–20 m intervals at one buffer area. Along transects the sampling plots were located at 2.5–10 m intervals, depending on the width of the buffer.

2.2 N retention capacity of wetland buffer areas

2.2.1 Artificial additions of N

To study the capacity of buffer areas to reduce N export in forested areas and to find out the main factors that contribute to the N retention capacity, we first used large artificial loadings of N in the form of ammonium nitrate (NH_4NO_3-N). The NH_4NO_3-N solution was added to six buffer areas once or twice during a monitoring period of 4-6 years (Paper I). During the first addition in 2003, 2004 or 2005 1 kg of NH_4NO_3-N (50% ammonium-N,

Table 1. Background information of the studied buffer areas (BAs).

	Asusuo	Murtsuo	Kirvessuo	Tulilahti	Konilampi	Hirsikangas	Kallioneva	Vanneskorpi
Location	60°26'N, 23°38'E	61°01'N, 28°19'E	61°14'N, 25°16'E	63°01'N, 26°59'E	61°48'N, 24°17'E	64°04'N, 26°40'E	62°16'N, 23°48'E	61°51'N, 23°42'E
BA (ha)	0.20	0.21	0.12	0.09	0.12	1.01	1.03	0.80
Watershed area (ha)	87	107	133	50	8	90	21	40
BA (% of watershed area)	0.23	0.20	0.09	0.18	1–2	1.12	4.88	2.00
Length of BA (m)[a]	30	50	55	90	20	100	320	120
MMT January (°C)[b]	-4.8	-8.0	-7.2	-9.5	-8.5	-9.9	-8.4	-7.3
MMT July (°C)[b]	17.0	17.2	16.8	16.5	16.5	15.5	15.4	15.5
MAP (mm)[b]	710	630	562	610	650	630	630	680
Date of addition								
1st add	26 May 2003	12 June 2003	1 July 2003			7 June 2004	17 July 2003	6 June 2005
2nd add	6 May 2008	28 May 2008				27 May 2008	21 April 2008	13 May 2008
N added (kg)								
1st add	27.6	38.0	44.9			43.1	7.2	13.8
2nd add	51.6	51.6				51.6	51.6	51.6
Site description	Undrained mire	Drained peatland	Drained peatland	Paludified mineral soil	Drained peatland	Undrained mire	Undrained mire	Drained peatland
Site type[c,d]	Herb-rich sedge hardwood-spruce fen	*Vaccinium myrtillus* type	Herb-rich type	*Vaccinium vitis-idaea* type	*Vaccinium vitis-idaea* type	Low-sedge *S.papillosum* flark fen	Tall-sedge fen	*Vaccinium vitis-idaea* type
Stand description	*Betula pubescens* dominated	*Betula pubescens* dominated	*Picea abies* dominated	*Pinus sylvestris* dominated	*Pinus sylvestris* dominated	Treeless	Treeless	*Picea abies* dominated
Stand volume ($m^3 ha^{-1}$)	80	80	100	30	10	0	0	100
Peat depth	>1m	>1m	>1m	<0.1m	>2m	>1m	>1m	>1m

[a]Distance between water inflow point and outflow point of a buffer.
[b]MMT= Mean monthly temperature, MAP= Mean annual precipitation, according to Drebs et al. (2002).
[c]Site types for peatlands according to Heikurainen and Pakarinen (1982), for mineral soils according to Cajander (1926).
[d]Site type at the time of buffer construction.

Table 2. Soil chemical and physical characteristics of the experimental peatlands. Soil temperatures, soil water levels, soil water pH, DOC and N concentrations during growing season of 2007 (during 1996–1999 for Kirvessuo).

	Depth	Asusuo	Murtsuo	Kirvessuo	Konilampi	Hirsikangas	Kallioneva	Vanneskorpi
Soil temperature (°C)	5cm	13.3±0.31	14.7±0.34	-	15.3±0.57	13.3±0.53	12.5±0.49	11.3±0.49
	30 cm	12.7±0.31	13.7±0.23	-	12.5±0.58	12.6±0.38	11.3±0.41	10.0±0.44
Peat								
BD (g cm^{-3})	0–7.5 cm	0.13±0.02	0.25±0.02	0.14	0.08±0.02	0.12±0.02	0.08±0.01	0.35±0.12
	7.5–15 cm	0.58±0.13	0.14±0.00		0.08±0.01	0.15±0.03	0.07±0.01	0.15±0.02
pH	0–7.5 cm	4.6±0.1	5.1±0.0	5.0	4.0±0.1	5.3±0.1	5.1±0.0	4.9±0.1
	7.5–15 cm	4.7±0.0	5.2±0.0	-	4.0±0.1	5.1±0.1	4.9±0.1	4.6±0.1
C (%)[a]	0–7.5 cm	25.9±5.8	19.6±2.8	50.9	53.8±0.41	25.3±3.7	37.0±3.9	21.3±9.1
	7.5–15 cm	6.0±1.5	37.6±1.5	-	54.7±0.83	25.4±6.0	40.0±2.8	37.3±6.32
N (%)[a]	0–7.5 cm	1.2±0.2	1.0±0.1	1.9	1.8±0.0	1.0±0.2	1.4±0.1	0.8±0.3
	7.5–15 cm	0.3±0.1	2.0±0.1	-	1.7±0.1	1.1±0.4	1.4±0.2	1.3±0.3
P_{tot} (mg kg^{-1}) [b]	0–7.5 cm	873±174	767±49	905	909±23	817±62	1066±47	900±51
	7.5–15 cm	416±25	1138±38	-	855±41	701±61	994±74	767±49
Ca (mg kg^{-1}) [b]	0–7.5 cm	2356±380	4865±261	7710	3080±351	3518±415	4568±16	5412±497
	7.5–15 cm	1768±116	6608±58	-	1565±279	3214±376	4270±26	4840±361
Mg (mg kg^{-1}) [b]	0–7.5 cm	2426±492	6330±457	1350	652±53	2308±161	3801±70	5384±1052
	7.5–15 cm	2590±174	3308±324	-	391±57	2294±321	3129±55	3468±879
K (mg kg^{-1}) [b]	0–7.5 cm	2034±146	3768±266	454	649±56	1464±165	2779±32	2814±439
	7.5–15 cm	1188±92	1933±211	-	335±43	1034±174	1830±25	1697±454
CEC (mmol kg^{-1}) [c]	0–15 cm	90±23	290±9	455	173±23	129±30	253±23	196±46
Water								
pH[c]		5.6±0.1	5.9±0.1	6.0±0.1	4.2±0.2	5.6±0.0	5.5±0.1	5.1±0.1
DOC (mg l^{-1})[d]		22.5±2.6	31.1±1.0	26.9±1.4	65.1±4.8	28.3±1.0	29.4±1.0	35.8±2.0
N_{tot} (mg l^{-1})[e]		0.53±0.04	1.41±0.35	0.56±0.02	2.73±0.82	0.57±0.03	1.27±0.02	0.94±0.08
Water table level (cm)		2.3±1.0	3.6±0.7	-	-14.8±2.1	12.1±0.7	-0.9±0.6	-8.9±1.1

Methods used: [a]LECO CHN-1000 analyzer (ISO 10694, ISO 13878), [b]Wet digestion in HNO_3/HCL, ICP/AES, [c](SFS 3021), [d]TOC-5000 analyzer (SFS-EN 1484), [e]Lachat Quickchem 8000 FIA analysator (SFS-EN ISO 11905-1).

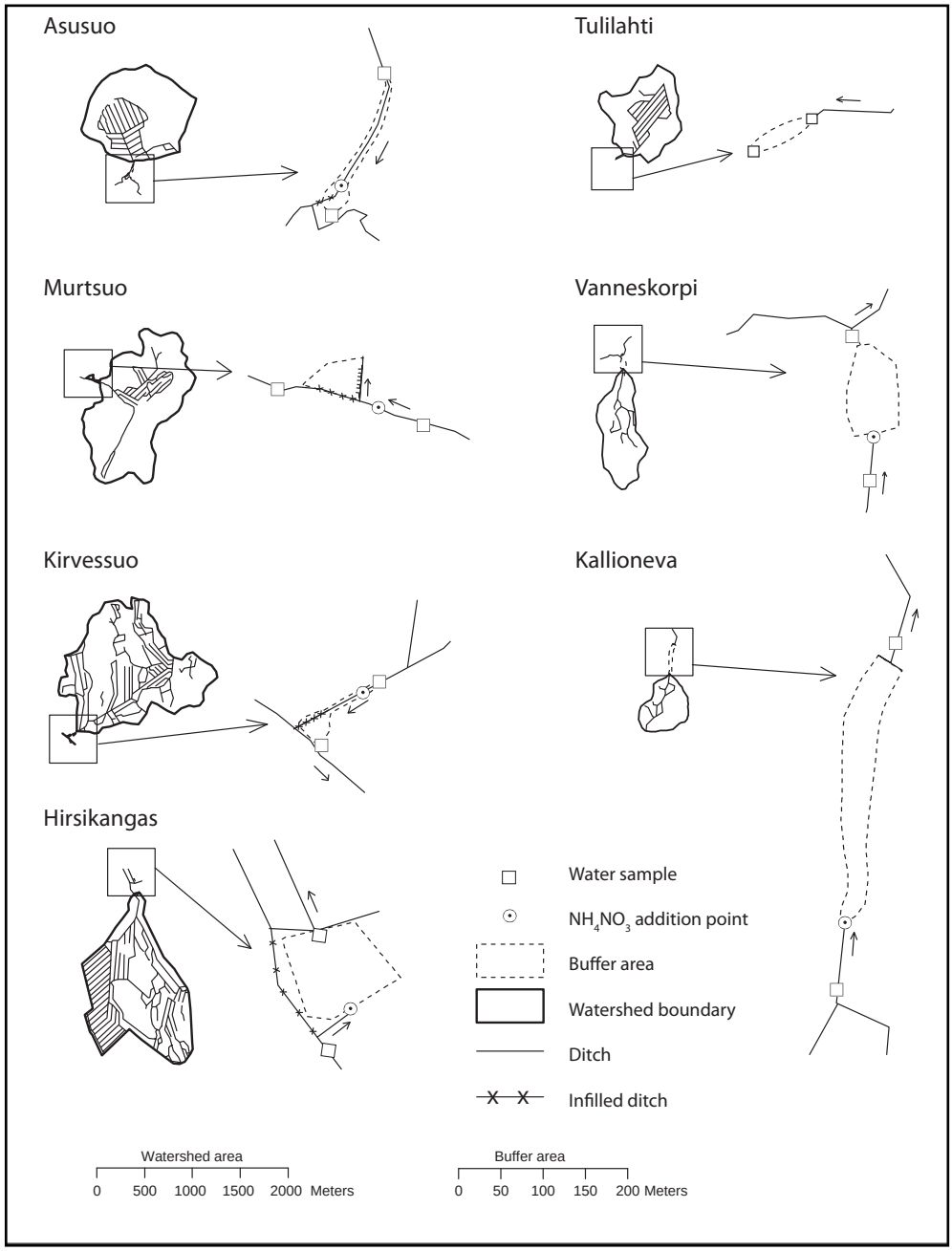

Figure 3. The catchment areas and experimental design of seven studied buffer areas. The Konilampi buffer area is presented in Silvan et al. (2004a).

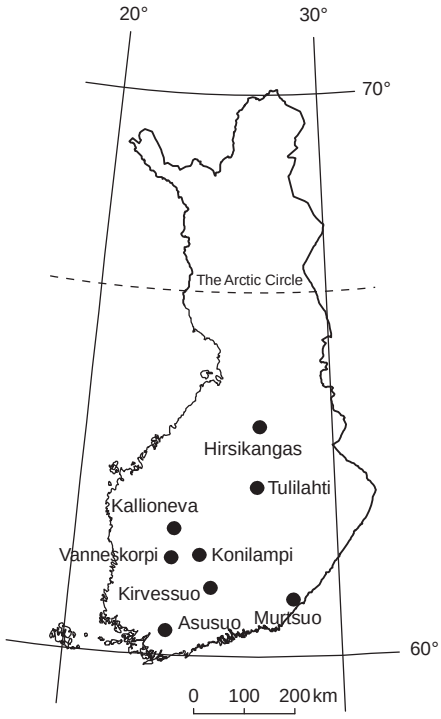

Figure 4. Location of the study sites in southern and central Finland.

50% nitrate-N) per one hectare of catchment area were added (7.0–374.0 kg per ha^{-1} of the buffer area), and in the second addition in 2008, each study area received a total of 51.6 kg of NH_4NO_3-N (50.1–258.0 kg per ha^{-1} of the buffer area). Each of the two additions lasted for four days. The daily NH_4NO_3-N input dose was dissolved in some local runoff water in a 0.2 m^3 (first addition) or 1.0 m^3 (second addition) PVC tank and the solution was then allowed to trickle into the runoff during a period of about 24 h.

At each of the six buffer areas, sampling of the inflow and outflow waters began on the same day as the first N addition in 2003–2005. Water samples were collected daily throughout the four-day addition period. After the first addition had ended, 6–18 samples per buffer area 1–3 times per month were collected during the year of N addition until the waters were ice-covered in late autumn. During the years with no N addition an average of seven samples 1–3 times per month were taken annually from each buffer area in the frost-free period. In the second addition period in 2008, water samples were collected daily during the four-day addition period, three times during the second week and twice during the third week after the addition. Samples were then collected weekly until the growing season ended. During autumn 2008, 1–2 samples per month were collected until the waters were ice-covered. The water samples were collected 1) upstream from the buffer area where the N addition had had no effect on the water quality and 2) from the outflow channel downstream from the buffer area. The total dissolved N, NO_3-N and NH_4-N concentrations were analyzed from filtered (0.45 µm membrane filters, Supor) water samples with a Lachat Quickchem 8000 FIA-analyser. Dissolved organic N (DON) concentrations for years 2007 and 2008 were calculated as the difference between total dissolved N and inorganic nitrogen. The analyses were done at the Finnish Forest Research Institute, according to the procedures described by Jarva and Tervahauta (1993).

The runoff in the buffer areas was recorded during each sampling occasion by measuring the height of the water level with an accuracy of 1 mm from the bottom of the V-notched weir. If there were no measurements made on the runoff, reference data of daily runoff from the nearby small research catchments operated by the Finnish Environment Institute were used.

The outflow of the added N from buffers during the N addition year(s) were calculated by the following formula:

$$N_{out} = \sum_{t=t_0}^{t_N} \frac{(c_{ON,t} - c_{BN})}{10^6} Q_t \ , \qquad (1)$$

where N_{out} is the total outflow of added NH_4-N or NO_3-N from a buffer area during the observation period after the N addition (kg), $c_{ON,t}$ is the concentration of NH_4-N or NO_3-N (mg l^{-1}) in the outlet ditch below the buffer area at time t (d), c_{BN} is the average background NH_4-N or NO_3-N concentration (mg l^{-1}) calculated from the water samples collected upstream from the buffer area where the N addition has had no effect on the water quality, Q_t is the water flow (l d^{-1}), t_0 is the first day of observation period, and t_N is the last day of the observation period. To produce continuous daily water flow and concentration time series water flow values and inorganic N concentration values in the outflow below the buffer areas were interpolated for each day between the measurement occasions.

The total N retention capacity of the buffer areas during the N addition years were calculated from:

$$r_c = \left(1 - \frac{N_{out}}{N_{in}}\right)\% \qquad (2)$$

where r_c is the retention capacity (%), N_{in} is the nitrogen added to the inflow (kg) and N_{out} is the outflow of the added N (kg).

2.2.2 Reduction of NH_4-N transport after ditch network maintenance

A widely used approach to study the retention efficiency and the related processes under large nutrient loadings is an artificial addition of nutrient solutions into the water entering the buffer areas at a high and steady loading rate during a time period of few days or months (e.g. Silvan et al. 2005, Väänänen et al. 2008). However, the nutrient addition experiments are unlikely to closely simulate sporadically increased and long-lasting loadings that have been shown to occur, e.g., after forest harvesting, fertilization and ditch network maintenance (Binkley et al. 1998, Ahtiainen and Huttunen 1999, Joensuu et al. 2002). The pattern and duration of the loading may strongly affect nutrient retention efficiency of buffer areas and information is currently needed from areas where the increased loading originates from an actual forestry practice rather than an artificial nutrient addition.

We investigated the capacity of riparian buffer areas to reduce the ammonium (NH_4-N) export originating from ditch network maintenance areas in peatlands drained for forestry purposes. Samples from inflow and outflow waters of buffer areas were collected during

the snowfree season before and after ditch network maintenance operations at six buffer areas (Paper II). The sampling was started as soon as the buffer construction operations were finished. Ditch network maintenance operations (ditch cleaning and/or complementary ditching) were performed at the drainage areas above each buffer area one to three years after the buffer construction. The maintenance operations accounted for an area of about 16–65% of the catchment area.

Water samples were collected twice a week during spring and from weekly to biweekly during other seasons. The samples were taken either from the overflow of a V-notched weir or directly from the water flowing in the natural flow channel. In the laboratory, NH_4-N was analyzed from filtered (1.0 µm fibre-glass filters) water samples with a Tecaton FIA-analyzer according to Jarva and Tervahauta (1993).

The annual NH_4-N export above and below the buffer areas was calculated by first multiplying the monthly mean NH_4-N concentration with the monthly runoff, which was obtained using the data from the nearby research catchments of the Finnish Environment Institute. The monthly NH_4-N exports were then summed up to produce the annual export. The efficiency of the buffer areas in retaining NH_4-N was calculated by subtracting the annual ammonium export below the buffer area from the export above the buffer area.

2.3 Sampling for N_2O

A negative environmental impact with the use of peatland buffer areas in reducing nutrient transport may be the fact that they act as potential sources of greenhouse gases into the atmosphere. To assess whether peatland buffer areas would significantly contribute to the emissions of the greenhouse gas nitrous oxide (N_2O) from forested catchments, we measured the N_2O emissions using the closed chamber technique (Alm et al. 2007) and a large artificial N loading (Paper I and III). To study the potential for elevated N_2O emissions, an extensive study was made at five buffer areas during one growing season before (2007) and one after (2008) the N loading. During the extensive study, the emissions were measured from eight sampling positions **(Fig. 5)** using a sampling interval of 1–5 weeks. To obtain information on the spatial and temporal variation in N_2O emissions in a buffer area, and in order to be able to estimate the total emission of the N_2O (kg per buffer area), the artificial N addition was repeated in one of the five buffer areas in 2010 using more intensive sampling with 14 gas sampling positions **(Fig. 5)** and a sampling interval of 1–7 days (Paper III).

All gas samples were analyzed using gas chromatography (HP 6080 series II) in the laboratory of the Southern Unit of the Finnish Forest Research Institute. The fluxes of N_2O were calculated from a linear change in the gas concentration inside the chamber as a function of time (Martikainen et al. 1995, Nykänen et al. 1995, Regina et al. 1996, Maljanen et al. 2003).

During each gas sampling, the temperatures of the peat profile at 5 and 30 cm depths were gauged at each of the gas sampling points. The water table levels were measured manually from plastic tubes inserted next to the gas sampling points and soil water samples were collected from the same tubes into 250 ml PVC bottles. The NH_4-N and NO_3-N concentrations in the water samples were analysed as described in Paper III.

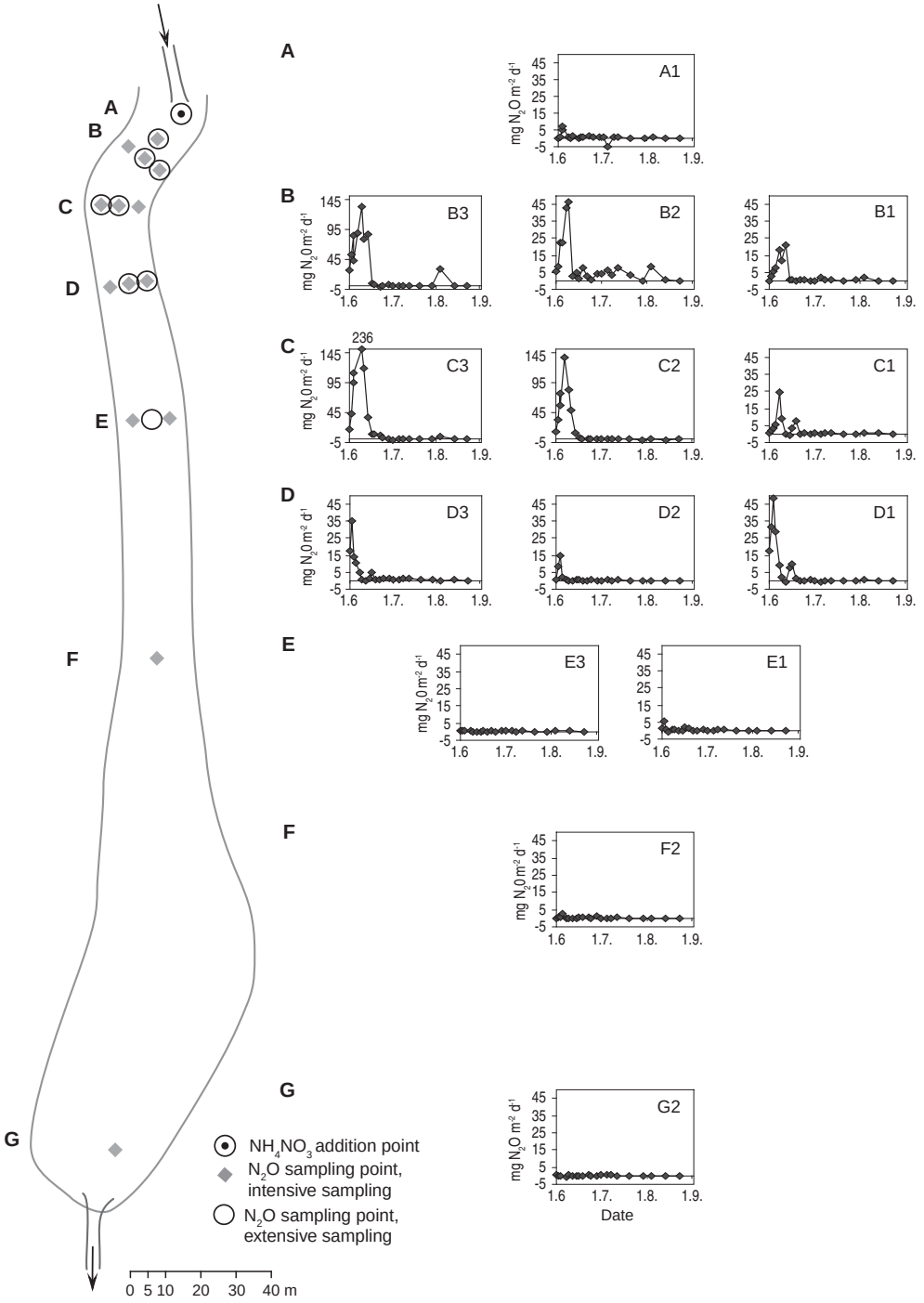

Figure 5. Layout of the experimental design for extensive and intensive sampling at Kallioneva buffer area. Emissions of N_2O at each of the 14 sampling points after the N addition in 2010 are provided in the associated graphs.

2.4 Sampling for CH_4

Methane (CH_4) is a powerful greenhouse gas also involved in the degradation of the ozone layer (Khalil 1999). We addressed how restoration of peatlands for use as buffer areas affects CH_4 emissions and the CH_4-cycling microbial populations. The CH_4 emissions and the contributing factors were measured at six buffer areas during one growing season (2007). Eight sampling points along four or five sampling lines were used for CH_4 sampling, and the sampling interval was one to five weeks. The gas sampling for CH_4 was made using the same procedure as for N_2O (Paper III and IV).

To analyze the microbial communities and the physical and chemical characteristics of the peat, peat samples were taken next to the eight gas sampling positions at each buffer area in August 2007. Peat monoliths (3×4×15 cm) down to a depth of 15 cm were divided into 0–7.5 cm and 7.5–15 cm sections. The sampling depth was 0–7.5 cm for methanotrophic bacteria and 7.5–15 cm for methanogenic archaea, whereas the basic chemical and physical characteristics of the peat were analyzed from both depths (Paper IV).

The microbial communities were compared by terminal restriction fragment length polymorphism (*mcrA* for methanogenic archaea), denaturing gradient gel electrophoresis (*pmoA* for methanotrophic bacteria) and sequencing (Paper IV).

2.5 Inventories for vegetation composition

One important component in the nutrient retention in peatland buffer areas is the nutrient accumulation by the living vegetation (Huttunen et al. 1996, Bedard-Haughn et al. 2003, Silvan et al. 2004a). Besides an increasing plant biomass (Aerts et al. 1995, Berendse et al. 2001), large nutrient loadings affect the buffer vegetation dynamics by inducing changes in the plant species composition (Heikkinen et al. 1994, Aerts et al. 1995, Huttunen et al. 1996, Bowman and Bilbrough 2001). However, such changes in vegetation composition dynamics may be an undesirable phenomenon in natural mires, especially when they belong to endangered and rare mire types.

We studied the vegetation composition dynamics in peatlands used as buffer areas in forested catchments. The plant species composition was recorded three times at three buffer areas between 1996 and 2009 (Paper V). The first vegetation inventory was done at the time of buffer construction, which was in 1996 or 2000. The second inventory was 4–5 years after the first one and the last vegetation inventory was performed in 2009 at all three buffer areas. The field and bottom layer vegetation was determined visually as percentage cover of each species (scale 0, 0.1, 0.2, 0.3, 0.5, 1, 2, 3...98, 99, 100%). Tree and shrub saplings of <50 cm in height were included in the study.

Global Non-Metric Multi-Dimensional Scaling GNMDS (Oksanen 2003, Venables et al. 2009) was used to study the changes in vegetation composition in the buffer areas. The GNMDS analyses were done separately for the upstream and downstream parts of the buffer areas (Paper V, Fig. 1) and also for the hummock surfaces and the lawn-level vegetation (including flarks). The vegetation inventory plots having a hummock or lawn-level cover of >70% were classified as hummock or lawn-level surfaces, and those plots having neither a hummock nor a lawn-level cover of >70% were discarded.

2.6 Statistical analyses

The Spearman correlation was used to study the relationship between the nitrogen retention capacity and the following factors (Paper I): the N loading, the relative size and the length of the buffer area, the average hydrological loading during five and seven days after N addition and the daily water flow during five and seven days after N addition.

The statistical significance of the measured changes in the NH_4-N concentrations before and after ditch network maintenance and between the inlet and outlet of the buffer areas were calculated using the non-parametric Mann-Whitney-Wilcoxon test, with a risk level of 0.05 (Paper II). The factors behind the variation in the annual NH_4-N retention efficiencies of buffer areas (i.e. the difference in the annual NH_4-N export between the inlet and outlet of the buffer areas) were analyzed by mixed linear regression models in order to account for autocorrelation between repeated measurements (McCulloch and Searle 2001). In the mixed models, two hierarchical levels of variation in the datasets were identified: a) between the buffer areas, and b) within the buffer areas between the measurement occasions. The tested explanatory variables were the buffer size (ha), the relative buffer size (% of catchment area), the buffer length (m), the coverage of buffer bottom and field layer vegetation (%), the volume of buffer tree stand (m^3 ha^{-1}), the soil bulk density (g cm^{-3}), the soil CEC (mmol kg^{-1}), the water flow to the buffer area from the upstream catchment (m^3 a^{-1}) and the NH_4-N loading to the buffer area (kg a^{-1}).

The differences in N_2O emissions between sampling occasions and buffer areas during 2007–2008 were tested using repeated measures ANOVA, where the buffer areas were used as a grouping factor and the sampling occasions as a repeated factor (Paper III). A Spearman correlation test was used to examine the dependence between N_2O emissions and the NH_4-N and NO_3-N concentrations in the soil water, the water table levels and the average soil temperatures (Paper III).

The methane emissions on the restored buffer areas and the buffer areas constructed on natural peatlands were compared using nested ANOVA on log-transformed data (Paper IV). Communities of the different sites were compared by analysis of similarity (ANOSIM) using Bray-Curtis (methanogens) or Dice (methanotrophs) distance measure and significance assessment by 10000 permutations. Within-site community variation of methanogens and methanotrophs was measured as multivariate dispersion by calculating distances from a centroid for replicates within each site with the PERMDISP2 program using Bray-Curtis (methanogens) or Dice (methanotrophs) dissimilarities (Anderson 2006). The distances of the restored buffer areas and the buffer areas constructed on natural peatlands were then compared using nested ANOVA in SPSS. Multivariate analyses were used to explore the variation of methanotrophic (PCA) and methanogenic communities (DCA), vegetation (DCA) and buffer area peat chemistry (PCA) and to link the variation to environmental variables (DCA, PCA) (Paper IV).

To study whether the mean coverages of plant species differed significantly between the three vegetation inventories, a non-parametric Friedman test with a risk level of 0.05 was used (Paper V). Multivariate analyses were used to study the changes in vegetation composition in the buffer areas (GNMDS, see Paper V) during 9–13 years after the buffer construction.

3 RESULTS

3.1 Nitrogen retention by wetland buffer areas

3.1.1 N retention after large short-term N loading

To study the N retention capacity of buffer areas on forested catchments, and to define the main factors that contribute to the N retention capacity, we added ammonium nitrate (NH_4NO_3-N) solution into the inflow waters of six buffer areas once (one area) or twice (five areas) during a monitoring period lasting 4–6 years. Samples for inflow and outflow waters of the buffer areas were collected before and after the N additions. The three largest buffer areas retained NO_3-N almost completely, with retention capacities of 93.1–100.0% **(Table 3).** The NH_4-N retention capacity in the three largest buffer areas was also very high, 93.3–99.9%, with the exception of one buffer area during the first addition year in 2003 (68.7%). Two of the three small buffer areas showed large capability to retain the added N with retention capacities of 58.0–86.8% for NO_3-N, and 73.7–89.2% for NH_4-N, respectively. One small buffer area had significantly lower retention efficiencies compared with the other five buffer areas; it retained only 10.2 and 15.3% of the added NO_3-N and 16.9% and 7.5% of added NH_4-N, after the two artificial N additions. Under conditions where inorganic N supply is high, such as here, some part of inorganic N may be transformed into the organic forms, and then be transported as organic N in the aquatic ecosystem (Murphy et al. 2000). However, our results did not indicate increased export of added inorganic N as DON (**Fig. 6**) except for that one sample from Asusuo showed high DON concentration.

Table 3. Mean water flow and hydrological loading to buffer areas during five days after starting the N addition and the N retention capacity of studied six buffer areas. DNM= Ditch network maintenance.

Buffer area	Year of addition	Hydrological loading ($mm\ d^{-1}$)	NH_4-N retention		NO_3-N retention	
			(kg)	(% of added)	(kg)	(% of added)
Asusuo	2003	89.9	2.3	16.9	2.1	15.3
	2008	51.7	1.9	7.5	2.6	10.2
Murtsuo	2003	85.9	14.0	73.7	11.0	58.0
	2008	55.8	22.2	85.9	22.3	86.6
Kirvessuo	2003	65.9	20.0	89.2	19.5	86.8
Hirsikangas	2004	3.2	21.4	99.1	21.2	98.6
	2008	3.1	25.8	100.0	25.6	99.3
Kallioneva	2003	1.5	2.5	68.7	3.6	100.0
	2008	20.6	25.8	100.0	25.8	100.0
Vanneskorpi	2005	7.4	6.9	99.7	6.8	98.1
	2008	2.2	25.7	99.5	24.0	93.1

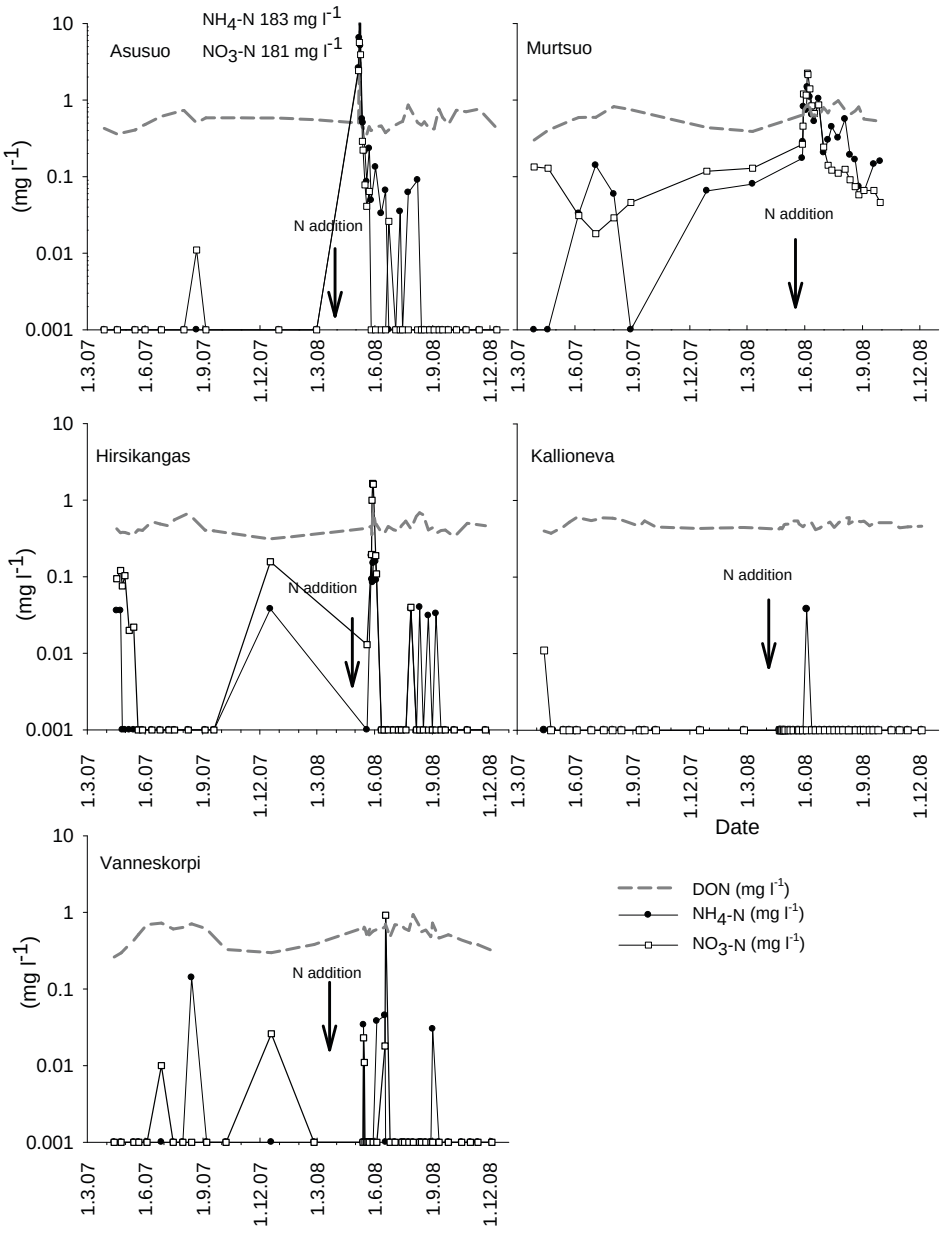

Figure 6. After the artificial additions of N in 2008, the concentrations of NH_4-N and NO_3-N in the outflow waters increased clearly above the levels before N addition in 2007 at three out of five buffer areas. The DON concentrations did not show increasing trends on account of the additions.

The Spearman correlation analyses showed that the NO_3-N retention was strongly and positively correlated with the length (r=0.92, *p*=0.000) and relative size (r=0.75, *p*=0.008) of the buffer area, but the NH_4-N retention had a strong positive correlation only with the buffer area length (r=0.65, *p*=0.032) (paper I). The N loading (kg per ha^{-1} of buffer area) correlated negatively with the retention efficiency of NO_3-N (r=-0.78, *p*=0.005). Similarly, the hydraulic loading (mm d^{-1}) had a significant negative correlation with the retention capacity of NO_3-N (r=-0.73, *p*=0.010).

3.1.2 NH_4-N retention after ditch network maintenance operations

The capacity of buffer areas to reduce nutrient loadings from discharge waters has been investigated mostly by using artificial nutrient additions and information is needed from areas where the increased export originates from an actual forestry practice. We studied the capacity of six peatland buffer areas to reduce the ammonium (NH_4-N) export originating from ditch network maintenance areas on forested catchments. At the large buffer areas (>1%), the retention capacity varied between 69.9 and 100.0% of the NH_4-N loading, 1–3

Table 4. Mean annual water flow and NH_4-N loading into buffer areas before and after ditch network maintenance (DNM) and NH_4-N retention efficiency of the studied buffer areas.

Buffer area	Years after addition	Water flow (m^3 a^{-1})	NH_4-N loading (kg a^{-1})	NH_4-N retention	
				(kg a^{-1})	(% of added)
Asusuo	-1	309 190	0.8	-0.5	-54.5
	+1	286 370	0.7	-1.4	-196.0
	+2	306 670	0.2	-0.0	-11.9
	+3	346 650	1.0	0.5	50.0
	+4	358 240	1.2	-0.5	-41.7
Murtsuo	-1	207 850	2.6	-2.2	-84.6
	+1	242 670	39.5	4.1	10.4
	+2	244 610	30.3	10.3	34.0
	+3	434 990	34.5	6.7	19.4
	+4	268 400	33.2	19.2	57.8
Kirvessuo	-3	264 590	2.3	0.6	26.1
	-2	443 470	5.7	1.8	31.6
	-1	350 050	8.1	7.5	92.6
	+1	394 300	57.7	12.2	21.1
	+2	300 030	54.2	21.5	39.7
Tulilahti	-1	130 020	0.6	-0.5	-83.3
	+1	141 510	8.2	6.1	74.4
	+2	125 690	10.9	6.4	58.7
	+3	109 410	9.8	7.4	75.5
Hirsikangas	-1	266 730	2.6	-0.3	-11.5
	+1	238 460	5.1	4.0	78.4
	+2	254 160	11.3	7.9	69.9
Kallioneva	-1	83 400	2.1	1.9	90.5
	+1	55 430	6.3	6.3	100.0
	+2	76 460	7.6	7.3	96.1
	+3	80 910	3.2	2.8	87.5

years following ditch network maintenance **(Table 4).** At the small-sized buffer areas (<0.3%), large variation was found in the retention efficiencies, as there was no retention for one area, but one buffer area had an annual retention efficiency of 58.7–75.5%. The retention in kg a^{-1} was 2.8–7.9 kg for the large buffer areas and -1.4–21.5 kg for the small buffer areas.

In the mixed model, the rate of NH_4-N loading into the buffer area covered alone about 68% of the variation in NH_4-N retention efficiency (Paper II, Table 3). Adding the buffer length or the water flow as explanatory variables into the model increased the goodness-of-fit of the model up to about 73%, and the retention of NH_4-N was non-linearly and positively related to the buffer length and negatively to the volume of water discharging to the area (Paper II, Figs. 3 and 4). The bottom layer or field layer vegetation cover, the buffer tree stand volume, or the soil characteristics (bulk density, CEC) of the buffer did not increase the goodness-of-fit of the models. Even more NH_4-N was released than retained in the buffer areas, when the NH_4-N loading was low. Simulations with the models constructed in paper II showed that the retention was very low or negative in conditions, when the annual NH_4-N loading was less than 1 kg a^{-1} and the annual water discharge into a buffer concurrently more than 300 000 m^3 a^{-1}. Negligible or negative retention occurred also under such circumstances, where the NH_4-N loading was below 10 kg a^{-1} and the length of the buffer area less than 40 m.

3.2 N_2O emissions before and after large N loadings

To investigate the potential for elevated N_2O emissions from peatland buffer areas, the N_2O emissions were studied using the closed chamber technique (Alm et al. 2007) and a large artificial N loading. First, an extensive study was made during one growing season before (2007) and one after (2008) the N loading to measure the emissions from eight sampling positions at five buffer areas using a sampling interval of 1–5 weeks (**Fig. 5**). Then the addition was repeated more intensively in 2010 at one area using 14 sampling positions and a sampling frequency of 1–7 days.

In 2007, the average N_2O emissions in the studied buffer areas varied from -0.33 to 0.14 mg m^{-2} d^{-1}, and after the N addition in 2008, they varied from 0.99 to 5.30 mg m^{-2} d^{-1} (Paper III, Fig. 3). At two buffer areas the N_2O emissions increased to a level of about 19 mg m^{-2} d^{-1} three days after the start of the N addition. One week later, the mean N_2O emissions had decreased to a level of about 3.80 mg m^{-2} d^{-1} at both buffers. At the other three buffers, the mean N_2O emissions were 0.98, 1.20 and 2.20 mg m^{-2} d^{-1} at the end of the N addition week, respectively. At four of the five sites, the highest emissions were measured within one and two weeks after the start of the N addition, but at one area the emissions peaked not until after two months. The emissions returned to pre-addition levels in about 9–15 weeks after the N addition.

The emissions of N_2O differed significantly with time (p<0.001) and the interaction between sampling time and experimental site was also significant (p=0.004) according to repeated measures ANOVA. The correlation analyses showed that during the high N loadings, the N_2O emissions correlated positively with the concentrations of NO_3-N (r=0.60, p<0.001) and NH_4-N (r=0.64, p<0.001) in the soil water and the rising water table level (r=0.37, p=0.018).

During the intensive sampling period in one of the five buffer areas in 2010, the N_2O emissions peaked at the end of the addition week being on an average 40 mg m^{-2} d^{-1}, about

300 times higher than the mean emission before N addition. The mean emissions were 19 mg m^{-2} d^{-1} at the end of the second week and 2.4 mg m^{-2} d^{-1} at the end of the third week following the addition. After that, the mean N_2O emissions remained below 1.0 mg m^{-2} d^{-1}, and returned to pre-addition level in about seven weeks. However, the emissions increased temporarily about ten weeks after the N addition, being about 3.3 mg m^{-2} d^{-1}.

High emissions of nearly 6.0 mg m^{-2} d^{-1} were found temporarily at 80 meters from the N addition point in the area of intensive sampling; however, the largest N_2O losses occurred within an area of 20 meters (**Fig. 5**). The emissions at the sampling points located at 160 and 320 meters did not increase much above 1.0 mg m^{-2} d^{-1}. The overall loss of N_2O during 2010 was calculated to be about 0.74 kg or 0.15 kg ha^{-1}.

3.3 CH_4 emissions from buffer areas constructed on natural and restored peatlands

The most important natural sources of the greenhouse gas methane (CH_4) are the methane-producing bacteria in swamps and wetlands, including peat-forming mires. To address, how restoration of a peatland ecosystem affects CH_4 emissions and CH_4-cycling microbial communities on a buffer area, we compared CH_4 fluxes and the key microbial populations in CH_4 cycling, methanogens and methanotrophs, in buffer areas constructed on natural mires and restored and rewetted peatlands.

The mean CH_4 emissions during the growing season were significantly higher from the buffer areas constructed on natural mires, being about 140–710 mg m^{-2} d^{-1}, than from the restored sites, where they were about 3–11 mg m^{-2} d^{-1} (p=0.001) **(Fig. 7)**. Two restored

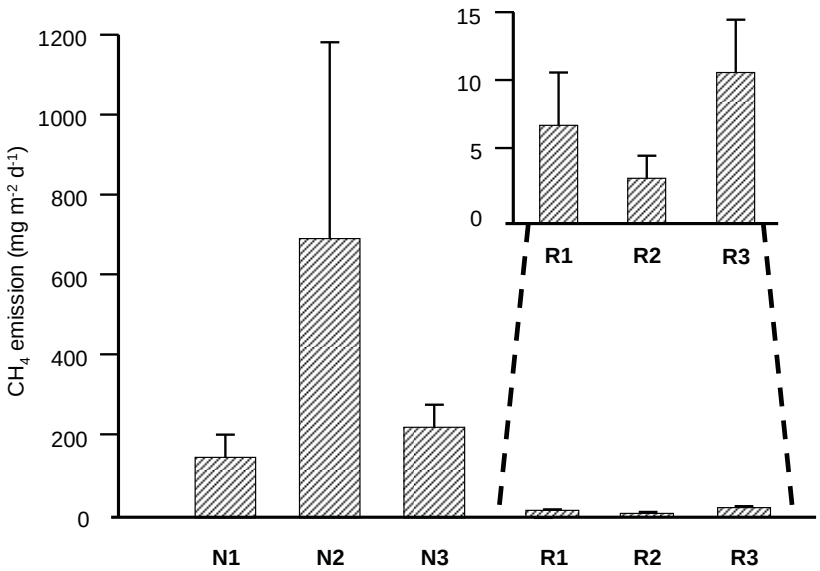

Figure 7. Methane fluxes of restored (R) peatland buffer areas and buffer areas constructed on natural (N) mires. The columns represent growing season averages (+ standard error) of eight sampling points per site. The insert shows the restored sites in more detail.

buffer areas even acted occasionally as sinks for CH_4. At the buffer areas constructed on natural mires, the highest emissions were measured in late July or August.

Different buffer areas shared similar methanogens (Methanobacteriaceae, Methanosarcinaceae and *Methanoregula*) but differences could be found in the heterogeneity of the communities. The communities were fairly homogenous within the buffer areas constructed on natural mires, while the community heterogeneity was higher at the restored buffer areas and the communities were also more widely scattered (Paper IV, Fig. 3). The community heterogeneity correlated negatively with the CH_4 emission ($r=-0.97$, $p=0.001$). Both methanotroph and methanogen communities varied with pH, DOC and a cation gradient of Ca–K–Mg. These factors did not differentiate the buffer areas based on restoration status but separated individual sites from the others (Paper IV, Fig. 4).

The methanotroph communities were nearly identical between the restored buffer areas and the buffer areas constructed on natural mires and consisted of type II methanotrophs (*Methylocystis*). Four out of six buffer areas showed identical banding patters (Paper IV, Fig. 4b), and the different sites were not differentiated based on the community composition (Paper IV, ANOSIM $r=0.06$ $p=0.025$) or multivariate dispersion (Paper IV $p=0.730$, Fig. 3). However, the buffer area with the highest water table level among the sites and the buffer area with the lowest water table level differed from the other sites by displaying only one or two of the common bands. The divergence of these sites could be explained by the variation of the peat chemistry and vegetation (Paper IV, Fig. 4b).

3.4 Vegetation changes in three buffer areas

The use of peatlands as buffer areas may induce undesirable changes in the plant species composition in the case of natural or nearly natural mires, especially when they represent endangered and rare mire types. The changes in vegetation composition were studied in two buffer areas constructed on natural mires (Asusuo, Hirsikangas) and one restored (Murtsuo) peatland buffer area.

Buffer areas constructed on natural mires

The upstream part of the Asusuo buffer was classified as a herb-rich sedge hardwood-spruce fen and the downstream part as a tall-sedge hardwood-spruce fen both at the time of buffer construction and 13 years later. Even though the site types did not change, clear changes occurred in the species composition, especially in the upstream parts of the Asusuo. At the time of the buffer construction in 1996, the field layer of the Asusuo buffer area was dominated by grasses and sedges, such as *Carex rostrata* and *Calamagrostis purpurea*, but herbs became the most dominant field layer plant group with an increase of from 6.7 to 20.4% in cover by the year 2009. Simultaneously, the coverage of *Lysimachia thyrsiflora* and *Menyanthes trifoliata* increased from 0.8 to 7.7% and 0.6 to 5.8%, respectively (Paper V, Table 4). However, *Calamagrostis purpurea* still maintained its state as the most common field layer species with coverage of 7.9% in 2009. The bryophyte layer of the Asusuo was dominated by *Sphagnum angustifolium* at the time of buffer construction in 1996, but its coverage had declined significantly by 2009, and concurrently, *Sphagnum riparium* increased in cover to become the dominant bryophyte.

The upstream part of the Hirsikangas buffer was classified as a low-sedge *Sphagnum*

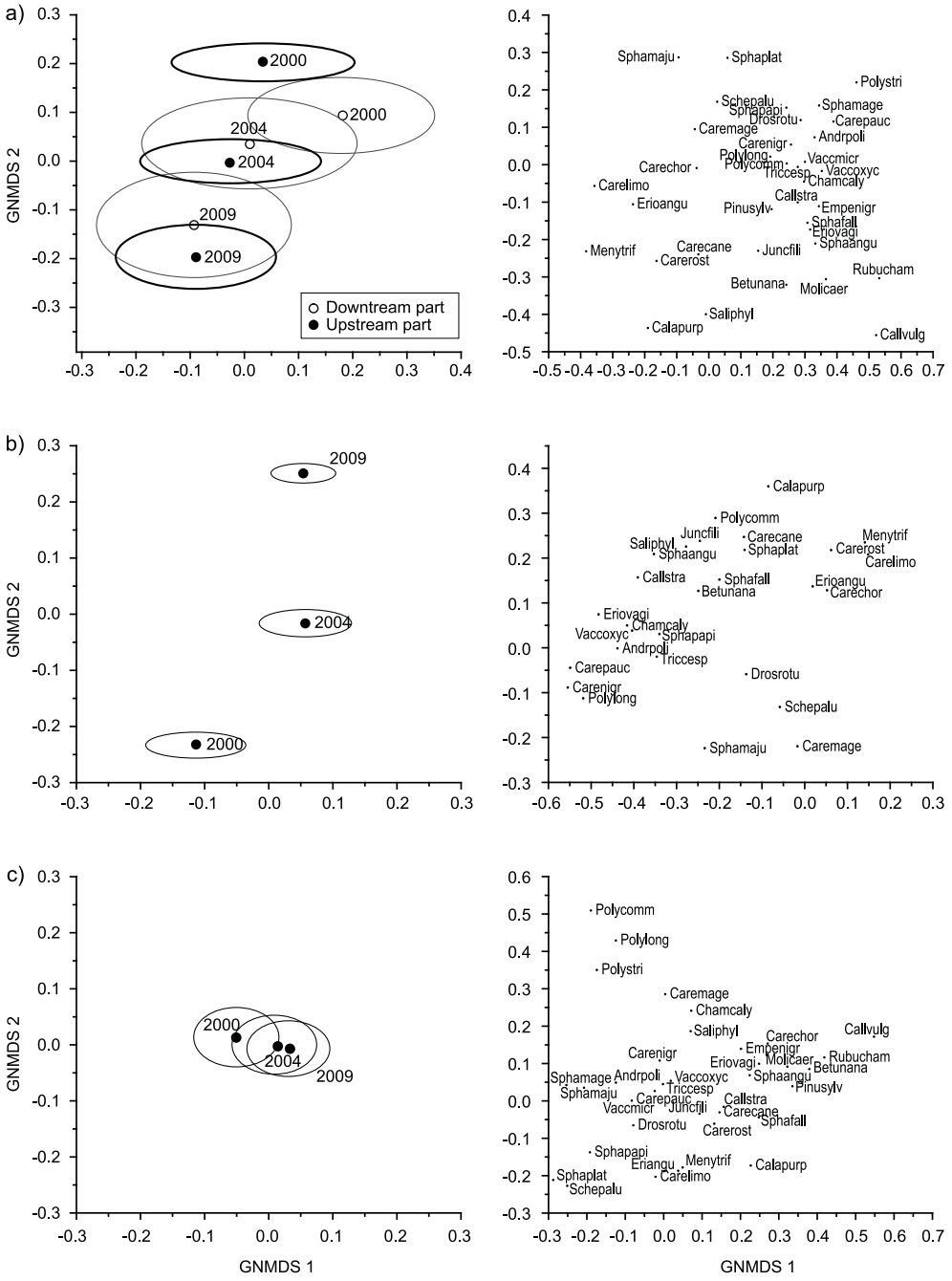

Figure 8. Three dimensional GNMDS ordination of the vegetation data at the Hirsikangas buffer area (n=52). Dimensions 1 and 2 are presented. Confidence intervals of 95% are presented as circles. (a) The upstream and the downstream parts are shown separately for each study year. In addition, the vegetation composition (b) at the lawn-level surfaces, including flarks, and (c) at the hummock surfaces are shown.

Figure 9. The vegetation at the upstream part of the Hirsikangas buffer area during the buffer construction in 2000 (Photo: Martti Vuollekoski).

Figure 10. The vegetation at the upstream part of the Hirsikangas buffer area in 2011 (Photo: Jorma Issakainen).

papillosum flark fen at the time of buffer construction in 2000, but as a herb-rich flark fen in 2009. The downstream part was classified as a flark fen both in 2000 and 2009. During the study, the coverage of grasses and sedges increased from 10.6% up to 36.7% (Paper V, Table 3) and the largest proportion of the growth was covered by *Carex rostrata* and *Carex limosa*, (Paper V, Table 4). Significant increases were also observed in the herbs group, where especially *Menyanthes trifoliata* increased in cover. The largest changes occurred at the upstream part of the buffer **(Figs. 8, 9 and 10)**; where e.g. the coverages of *Calamagrostis purpurea* and *M. trifoliata* increased up to ten-fold in some plots, and *Salix phylicifolia*, which was absent in 2000, appeared widespread in 2009 (Paper V, Table 4).

Restored peatland

The Murtsuo buffer area was classified as a *Vaccinium myrtillus*-type drained peatland forest at the time of the buffer construction in 1996, but as a herb-rich sedge hardwood-spruce fen in 2009, after having been used for 13 years as a buffer area. Between 1996 and 2001, the coverage of grasses and sedges first increased from 1.0 to 24.6%, but by 2009, the coverage had decreased to 12.0% (Paper V, Table 3). *Calamagrostis purpurea* had undergone the largest part of the changes, as its coverage increased from 0.7 to 15.5% between 1996 and 2001 and then decreased to 10.5% (Paper V, Table 4). During the study, the abundance of the shrubs and tree saplings decreased from 25.7 to 2.2% mainly due to declined coverages of *Picea abies* and *Salix aurita* (Paper V, Fig. 4).

The GNMDS analyses revealed that the vegetation composition at all three buffer areas had changed significantly during our study. At the Asusuo and Hirsikangas buffer areas, the vegetation near the water inflow point had changed more than the vegetation further downstream of the buffer areas, whereas the vegetation composition at the restored Murtsuo buffer area had changed substantially both at the upstream and the downstream parts (Paper V, Figs. 3, 4 and 5). The GNMDS analyses showed that the changes in vegetation composition had mostly occurred on the lawn-level microsites and no significant change was apparent in the plots representing the hummock microsites.

4 DISCUSSION

4.1 Capacity of buffer areas to retain inorganic N from through-flowing waters

Our studies (Papers I and II) showed that large buffer areas with a relative size of >1% of their catchment area were highly effective in retaining inorganic N from the through-flowing waters. The retention efficiencies for the large buffer areas after large artificial additions of ammonium nitrate, were >93% for both NO_3-N and NH_4-N, except for one area during the first N addition. After ditch network maintenance operations, the NH_4-N retention capacities for the large buffer areas were also high, varying from 70 to 100%.

Except for one buffer area, the small buffers with a relative size of <0.3% were also able to reduce inorganic N from the through-flowing waters effectively after the artificial N addition experiment, having the retention capacities of 58–89% for NH_4-N and NO_3-N. Larger variation in the retention efficiencies were found after ditch network maintenance operations, as the annual NH_4-N retention efficiencies at three of the four small buffer areas varied from 10 to 93%, during the 2–4 years following ditch network maintenance. Larger

variation in the retention efficiencies after ditch network maintenance compared with artificial N addition is probably related to a more varying N loading, as the rate of N loading into the buffer area may cover most of the variation in N retention efficiency (Paper II, Table 3).

One of the small buffer areas showed very low retention capacities both after the artificial N addition experiment, being <17% of added N, and after ditch network maintenance operations, when even more N was released than retained. The poor retention capacity after the N addition experiments was mainly a consequence of the insufficient length of the buffer, as the distance between the water inflow and outflow points of the buffer area was only 30 meters (>50 m for the other areas). This led to the formation of continuous flow channels across the buffer (Woltemade 2000, Ronkanen and Kløve 2009). After ditch network maintenance, the poor performance of that buffer area was also because the N loading was close to the background level of forested areas and no N retention may occur during such low loadings (Paper II). Given that the retention capacity for phosphorus (24%) was also poor in that same buffer area (Väänänen et al. 2008), the rationale of constructing such small and short buffers should be questioned.

The results agree with the previous findings, which suggest that the relative size of the buffer area is important in controlling the outflow of nutrients in forested catchments (Woltemade 2000, Nieminen et al. 2005, Väänänen et al. 2006, 2008). The large size is relevant because the size is directly related to the extent of the nutrient retaining sinks and also because the water residence time is likely to be longer in the large buffer areas compared with the smaller ones. A long water residence time enables a close contact between the nutrient-rich through-flow water and the nutrient sinks in soil and vegetation (Heikkinen et al. 1994, Sallantaus et al. 1998, Dosskey 2001, Väänänen et al. 2008).

However, also smaller buffer areas may be effective in reducing nitrogen loadings, as seen in our study. In that case, the length of the buffer may be sufficient or the hydrological loading so low that the water residence time is adequate for the efficient retention of N (Väänänen et al. 2008, Ronkanen and Kløve 2009). The length of the buffer is also important because the formation of continuous flow channels across the buffer more likely occurs in short and wide buffer areas, than in long and narrow buffers of the same size (Woltemade 2000, Ronkanen and Kløve 2009) **(Fig. 11).**

Our results were in accordance with those of Silvan et al. (2003, 2004a) and Väänänen et al. (2008) concluding that especially large buffer areas are able to effectively reduce nutrient transport. The results supported the previous findings also in that the retention efficiency may decrease when nutrient loadings increase to a level where nutrient sinks in soil and vegetation may become saturated (Ronkanen and Kløve 2009) (Papers I and II). When the loadings are already near the background levels of forested areas, the buffer areas have little effect on the N transport or may even release N into the through-flow waters.

4.2 N_2O emissions from peatland buffer areas under large N loadings

The emissions of N_2O were studied using an artificial N loading. In order to study the potential for high N_2O emissions, we chose to use a significantly higher N loading (51.6 kg of N per buffer area as NH_4NO_3-N added during four days) than is likely to become realized under actual conditions in forested catchments. Our results showed that the emissions of N_2O from buffer areas before N addition were low, but they increased significantly at each buffer area after the addition (Paper III). The total loss of N_2O during

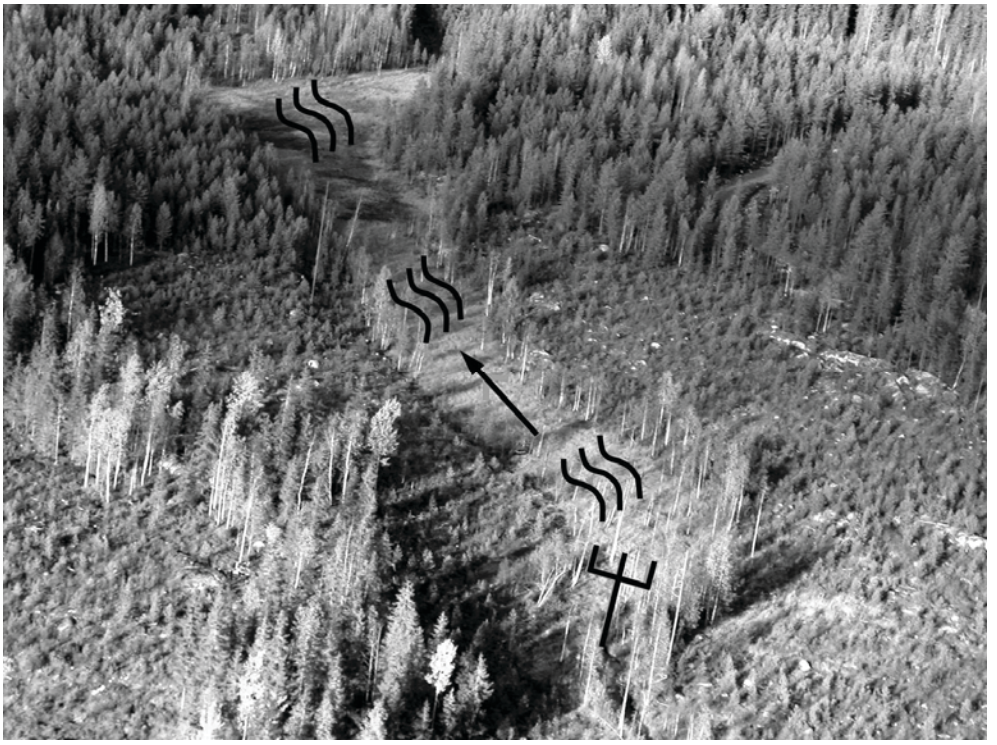

Figure 11. The relationship between the buffer length and the retention efficiency is probably explained by the fact that the probability of the formation of continuous flow channels across the buffer area is lower for long buffers than short and wide buffers of the same size. Also, the water residence time increases with the increasing length (Photo: Martti Vuollekoski).

the year of N addition was estimated to be 0.74 kg or 0.15 kg ha^{-1}. Despite the substantial increases, the emissions of N_2O remained lower than the annual emissions (0.12–3.50 kg ha^{-1} a^{-1}) reported by Regina et al. (1996) and the emissions (0.50–1.40 kg ha^{-1} a^{-1}) by Martikainen et al. (1995) for drained minerotrophic peatland forests. They were also substantially lower than the emissions from the peat soils drained for agricultural purposes in the boreal region (18–29 kg ha^{-1} a^{-1} in Regina et al. 1996; 36–81 kg ha^{-1} a^{-1} in Koops et al. 1997; 0.3–19 kg ha^{-1} a^{-1} in Regina et al. 2004). Thus, our study showed that although the emissions of N_2O increased after large N loadings, they remained smaller than from many other types of peat soils. It was concluded that the buffer areas in forested catchments should be viewed as a minor source of atmospheric N_2O.

The emissions of N_2O correlated positively with the concentrations of NO_3-N and NH_4-N in the soil water and the level of the water table. The effect of water table level is probably due to that a high water table level resulted in an effective dispersion of the added N and increased the availability of N in the soil water to denitrifying bacteria. If the hydrological conditions prevented an effective dispersion of the added N, the maximum emission levels and the overall N_2O losses remained lower. Smaller N_2O emissions under drier conditions are probably a consequence of closer contact between the through-flow

water and the vegetation and soil sinks of N, thus leading to lower concentrations of NO_3-N in the soil water (Paper III, Fig. 3). In a study by Silvan et al. (2004b), for instance, the vegetation was the main NO_3-N competitor for a denitrifying microbial population and a strong controller for N_2O emissions during the growing season. The emissions of N_2O after the N addition in 2008 were lower from the two restored buffer areas than for the three sites constructed on natural mires, again probably because the restored sites were drier.

4.3 The effect of restoration on the emissions of CH_4

The emissions of CH_4 were substantially lower from the three buffer areas restored 10–12 years earlier than from the three buffer areas constructed on natural mires. The results coincide with the previous studies showing that although restoration of peat soils increases CH_4 emissions, they may remain lower than the emissions from corresponding natural sites (Komulainen et al. 1998, Tuittila et al. 2000, Waddington and Day 2007). The mean emissions (140–710 mg m^{-2} d^{-1}) from our buffer areas constructed on natural peatlands were at the higher end of the range reported by Nykänen et al. (1998) and Whalen (2005) for pristine mires in the boreal region, and from our restored sites (3–11 mg m^{-2} d^{-1}) they were at the same level as for drained and forested peatlands in the same region (Roulet et al. 1993, Martikainen et al. 1995, Nykänen et al. 1998, Strack et al. 2004).

The lower CH_4 emissions in the restored peatland buffers were probably a result of the slow establishment of methanogens after the disturbance effect of the drainage (Tuittila et al. 2000). Although the different buffer areas shared similar methanogenic archaea, the community variation in the restored ones was significantly higher. The variation may reflect the presence of methanogens only in distinct microhabitats in drained oxygen-rich peat (Knorr et al. 2008), resulting in spatially heterogeneous communities when anoxic conditions return with restoration. However, the high spatial heterogeneity of methanogen communities in restored sites could also indicate incomplete restoration of the sites.

In contrast to methanogens, the methanotroph communities indicated no general or long-lasting response to drainage. As aerobes, methanotrophs may persist through drainage and restore more easily than methanogens, despite the diminished CH_4 supply, and therefore CH_4 oxidation is likely to be less affected by drainage than CH_4 production (Roulet et al. 1993). Instead, the factor behind variation in the methanotroph community structures was the site-specific differences in peat chemistry and vegetation (Paper IV, Fig. 4).

High water table and stable anoxic conditions have shown to be focal for CH_4 production (Andersen et al. 2006, Basiliko et al. 2007), thus restoration of the hydrology is essential for increased CH_4 emissions from restored peatland buffer areas. The formation of *Sphagnum* peat or development of herbaceous vegetation support the production of CH_4 by providing a good supply of labile carbon compounds needed for methanogenesis (Glatzel et al. 2004, Marinier et al. 2004, Waddington and Day 2007). Our study indicates that longer time scales than over 10–12 years may be required to restore the pristine-like CH_4-cycling microbial processes in rewetted peatland buffers.

4.4 Vegetation composition dynamics in peatland buffer areas

Peatland buffer areas are important means in reducing sediment and nutrient loading from discharge waters in a variety of landscapes; however, use of natural mires as buffer areas should be carefully considered as their use may induce unwanted changes in the plant species composition. The changes are associated with the different conditions in peatland buffer areas compared with the other types of peat soils; the water level is generally clearly above the soil level and the surface waters are in constant movement across the buffer. Also, the inputs of nutrients and sediment to the peatland buffer areas can be significantly larger than in other peat soils. In our study, the exposure of the buffer area vegetation to high hydrological, sediment and nutrient loading resulted in significant changes in the vegetation composition. The changes were most apparent in the upstream parts of buffers and wet lawn-level conditions, where the contact between the vegetation and the through-flow waters rich in nutrients and sediments were the closest. The vegetation in the downstream parts and the dry hummock surfaces did not change much.

The results were in accordance with the previous studies in that grasses and sedges, as well as herbs were generally favoured by the use of peatlands as buffer areas (Huttunen et al. 1996, Bowman and Bilbrough 2001, Hájek et al. 2006). The cover of grasses and sedges increased during the first 4–5 study years at all three buffer areas. However, the increasing trend appeared to be temporary at two buffers, as the coverages of the grasses and sedges then decreased after eight years. The enhanced growth of the grasses during the first five years was probably related to the decreased coverage and shading effect of shrubs and tree saplings, mainly *Salix aurita*, *S. phylicifolia* and *Picea abies*, which led to an emerging life space and an increased supply of light and nutrients. However, the competitive advantage for grasses was partly lost later. At one buffer, the disappearance was presumably associated with the gradually thickened sediment layers carried by the overland flow, which then weakened the living conditions for the grasses as well as for the other field and bottom layer species. At another buffer area, one probable reason for the disappearance of the grasses was the colonization of the herbs, which reduced free living space.

At the species level, the coverages of *Menyanthes trifoliata* and *Calamagrostis purpurea* increased the most. As found in previous studies from peatland buffer areas (Huttunen et al. 1996, Saari et al. 2010b), the nutrient-rich through-flowing waters favor the growth of these two mesotrophic lawn- or flark level species (Laine and Vasander 2005). At our buffer areas, their widespread occurrence at the time of the buffer construction may have promoted their further growth, since extensive nutritional amplitude has shown to enable better utilization of an increased supply of nutrients (Komulainen at al. 1998, Silvan et al. 2004a). Besides *Menyanthes trifoliata* and *Calamagrostis purpurea*, other species reflecting mesotrophy and wet conditions, like *Potentilla palustris*, *Carex limosa* and *Lysimachia thyrsiflora* (Hämet-Ahti et al. 1986, Bragazza and Gerdol 1999) increased in coverage.

Significant changes in vegetation composition were also observed at the upstream part of the buffer that was classified as a herb-rich sedge hardwood-spruce fen, a very rare and endangered mire site type in southern Finland (Laine and Vasander 2005). According to our results, the use of natural mires as buffer areas will result in significant changes in species composition and their widespread use as buffer areas should therefore be carefully considered.

5 CONCLUSIONS

In order to prevent the leaching of nutrients from forested areas, it is currently recommended that nutrient-rich drainage waters are conveyed to receiving water courses through wetland buffer areas. The use of buffer areas in forested catchments has been actively investigated during the last 15 years; but until now, the research has mainly concentrated on the reduction of sediment and P loadings, instead of N. However, previous studies indicate that many boreal waters may be N-limited, which makes the reduction of the N loadings increasingly important.

The results in our studies are in accordance with those regarding sediment and P, concluding that large buffer areas with size of >1% relative to catchment area upstream are effective in retaining nutrients and sediment from through-flowing waters. However, our studies indicate efficient nutrient removal also for smaller buffer areas. The key-factors controlling the retention efficiency are the size and shape of the buffer area, the hydrological loading and the rate of nutrient loading.

Although the use of wetland buffer areas can be seen as an effective water protection method, some undesirable environmental effects may be involved in the use, such as increased emissions of N_2O and CH_4 and changes in vegetation composition of the endangered site types. Our results show that although the N_2O emissions may increase temporarily after a large N loading into the buffer area, the buffer areas in forested catchments should be viewed as minor sources of N_2O. Our study indicated much lower CH_4 emissions for restored buffer areas than for buffers constructed on natural peatlands, probably because of the poor establishment of methanogens after drainage. Presently, several mire types are very rare or in an endangered state especially in southern Finland due to drainage. Our results indicate that the use of such mires as buffer areas can induce unwanted changes in their species composition, and therefore their use should be carefully considered. Instead, the construction of new peatland buffer areas should primarily be directed into restored peatland areas, where the changing vegetation is not an unwanted phenomenon.

6 PROPOSALS FOR FUTURE STUDIES

From the results of this thesis, some questions have been answered but there are still unanswered questions related with the use of peatland buffer areas. One question that is not fully understood is why restoration of some peat soils for use of buffer areas appears to lead to increased P losses, whereas on other sites the P exports will not increase after restoration. It may be that the P export increases especially from sites where a significant proportion of the P under drained conditions was adsorbed by iron hydroxides or oxides as this P is easily released upon water table rising and consequent redistribution and reduction of iron. Should the peatland sites have been fertilized with low release P fertilizers, it has been suggested that the rising of the water table may also increase the P exports from the undissolved fertilizer residues and the dissolved P bound in a labile form in the soil. The transport of P may be changed due to the water table rising also because the forest plant species typical for drained peatland forests have died and been replaced by mire species. The factors

behind increased P transport due to peatland restoration and rewetting should be a subject of future research.

It is recommended not to construct buffers on very flat areas, as the rising of the water table level in the buffer area results in the rising of the water table level also in the upstream forest area. The rising of the water table level in the forest area may have detrimental effects on tree growth. However, a sufficient slope for enabling the construction of a buffer area is not suggested in water protection guidelines for operational forestry. Also, the advantages (improved water purification) and disadvantages (decreased tree growth) of constructing buffers on flat areas have not been evaluated properly.

The inorganic forms of N, NO_3^- and NH_4^+, have dominated the discussion on N export in boreal environments, mainly because these forms are directly available for algal and microbial growth in aquatic ecosystems (Vitousek et al. 1997). In boreal forested catchments, however, the organic N generally constitutes the largest share of the N in the discharge waters (Lepistö et al. 1995, Kortelainen et al. 1997, Mattsson et al. 2005), and studies show that not only the inorganic forms of N, but also the organic forms can supply N nutrition to bacteria and phytoplankton (Seitzinger and Sanders 1997, Berman and Bronk 2003). The dynamics and cycle of organic N in peatland buffer areas is a subject for further studies.

More data is also needed on the nutrient retention in the buffer area vegetation and soil, as the previous investigations have indicated contradictory responses after increased nutrient availability. While about 70% of the N was retained in the vegetation biomass in a buffer area in central Finland (Silvan et al. 2004a), the retention of N in the vegetation biomass accounted for only 4% of the N retention in a buffer area in northern Finland (Huttunen et al. 1996). Also, while no significant increases were found in the N concentrations in plants after increased N availability in the study by Silvan et al. (2004a), the N concentrations in plants increased from 1.00 to 1.24% of dry weight in the study by Huttunen et al. (1996). The retention of nutrients in soil and vegetation has mainly been quantified by collecting samples before and after large nutrient loadings, which is problematic because the sampling positions can not be exactly the same. The use of marked isotopes of N and P could overcome that problem, but also their use has limitations. The ^{14}N tracer labelling approach may not be used in catchment-scale studies due to the cost of adding enough ^{14}N to gain sufficiently high ^{14}N enrichment in the soil and vegetation. For ^{32}P, long-term monitoring periods are not possible because the degradation rate of ^{32}P to ^{31}P with a half-life of 14 days lowers the radioactivity level below detectable limits. Nevertheless, further research is needed about the nutrient retention in the vegetation and soil in the peatland buffer areas.

This study showed that hydrological loading is one of the key-factors controlling the nutrient retention efficiency of the peatland buffer areas. Hydrological loading to the buffer area is governed by the size of the buffer relative to upstream catchment area, and large buffer areas are subjected to significantly lower hydraulic loadings than small buffer areas. In operational forestry, however, the shortage of suitable sites for buffers means that, there is a limited potential to adjust the size of the buffer area to match the size of the catchment. Thus, small buffer areas of <0.5 ha are as likely to be found in larger catchments (100 ha), where the hydrological loading is significantly greater, as they are in small catchments of a few tens of hectares, which have small loading appropriate to a small buffer.

One possible means to improve the nutrient retention efficiency of small buffers would be the use of the peak runoff control method (PCR). This means a set of control dams that would regulate the flow above the buffer areas (Marttila and Kløve 2009). With PCR, the

water flow is stored temporarily behind the dams in the ditch network during events of high runoff. As high hydrological loadings significantly decrease the nutrient retention efficiency of buffer areas (Paper I and II), integrated use of PCR and peatland buffer areas should be studied as a new tool in managing sediment and nutrient transport in forested catchments.

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